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HYDROGEOCHEMICAL, VEGETATIONAL AND MICROBIOLOGICAL EFFECTS OF A NATURAL AND A CONSTRUCTED WETLAND ON THE CONTROL OF ACID MINE DRAINAGE

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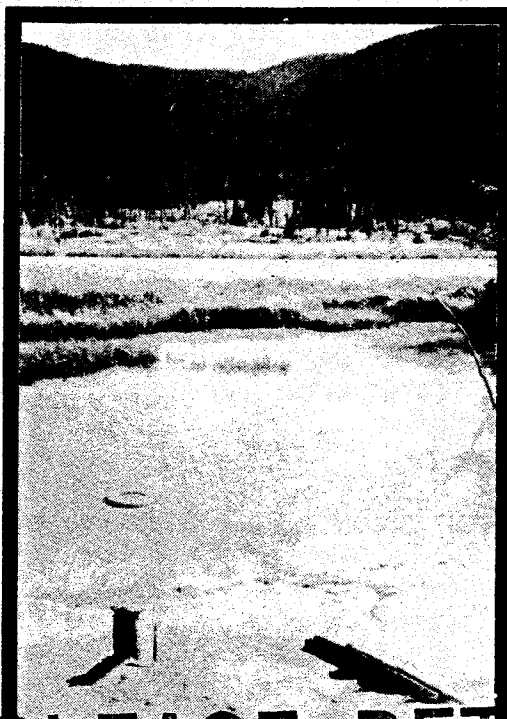
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PREFACE

Staff of the Reclamation Research Unit were responsible for investigation of the wetland hydrology, geochemistry, water quality and vegetation. Dr. Douglas J. Dollhopf (Soil Scientist) was the Principal Investigator, John D. Goering was the Geohydrologist, Robert B. Rennick was the Vegetation/Wildlife Scientist and Robert B. Morton was the Graduate Research Assistant (Geohydrologist). These investigators prepared Sections 1.0 through 9.0.

Staff from MSI Detoxification Incorporated (MDI) conducted the wetland microbiological investigation as reported in Section 10.0. Mr. Peter M. Jones was Project Manager, Dr. W. Kennedy Gauger was Senior Microbiologist, Dr. Jim B. Guckert was the Microbial Ecologist, Dr. Keith C. Cooksey was the Lipid Biochemist, Karen Bucklin was the Environmental Engineer, Rebecca Weed was the Geochemist and Margaret M. Lehman was the Toxicologist.

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Title and Subtitle Hydrochemical, Vegetational and Microbiological Effects of a Natural and a Constructed Wetland on the Control of Acid Mine Drainage	Report Date June 1988
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Abstract An evaluation of a natural wetland in western Montana, which has received acid mine drainage (AMD) from an abandoned lead/zinc mine for at least 40 years, indicated that the system is effective in removing iron from influent water, but is less effective in removing manganese and other metals. Approximately 550 metric tons of iron have been deposited in the wetland, while only about 1 metric ton of manganese has been deposited. Water velocity through the Swamp Gulch wetland was most accurately determined using the bromide tracer method, rather than measurements of hydraulic conductivity. Mean velocity was 2.6×10^{-2} cm/s (74 ft/day) and ranged from 3.1×10^{-1} to 6.2×10^{-4} cm/s. Dominant plant species included <u>Carex rostrata</u> , <u>Betula glandulosa</u> , <u>Salix boothii</u> and <u>Sphagnum tenellum</u> . Tissues of these plants exhibited elevated metal levels. Metal concentrations were generally well below phytotoxic levels, however, levels of cadmium in some plant tissues exceeded the maximum dietary intake level for domestic animals. This indicates a potential risk to animals whose diet contain large proportions of these plants. Anaerobic microorganisms, as well as abiotic processes are responsible for acid reduction in the Swamp Gulch wetland. Sulfate-reducing bacteria are functioning at the natural Swamp Gulch wetland, but not at the constructed Sand Coulee wetland. This is most probably a function of adequate food and energy sources at the natural wetland. These studies strongly suggest that poor performance by the constructed wetland is due to reduced microbial activity rather than its relatively small size. The microbial component of natural wetlands should be transferrable to artificially constructed systems.	
Keywords Acid Mine Drainage, Heavy Metals, Wetlands	

TABLE OF CONTENTS

	<u>PAGE</u>
Disclaimer/Acknowledgements/Preface.	i
Report Documentation Page.	ii
Table of Contents.	iii
List of Tables	vii
List of Figures.	ix
1.0 INTRODUCTION	1
1.1 ACID MINE DRAINAGE	1
1.2 STUDY OBJECTIVES	2
1.3 CONCLUSIONS.	2
<u>1.3.1 Wetland Hydrogeochemistry.</u>	2
<u>1.3.2 Wetland Vegetation/Wildlife.</u>	4
<u>1.3.3 Wetland Microbiology</u>	5
1.4 RECOMMENDATIONS.	6
<u>1.4.1 Wetland Hydrogeochemistry</u>	6
<u>1.4.2 Wetland Vegetation/Wildlife.</u>	7
<u>1.4.3 Wetland Microbiology</u>	7
2.0 NATURAL SYSTEMS FOR WATER POLLUTION CONTROL.	9
2.1 WETLAND TREATMENT SYSTEMS.	9
2.2 BIOSORPTION OF ELEMENTS BY PLANTS.	10
2.3 METALLOPHYTES AND LAND/WATER RECLAMATION	11
2.4 CHEMICAL PROCESSES OF ACID MINE DRAINAGE	11
<u>2.4.1 Acid Water Formation</u>	11
<u>2.4.2 Bacterial Mediation.</u>	11
<u>2.4.3 Influence of pH.</u>	11
<u>2.4.4 Pyrite Oxidation Control</u>	12
3.0 SITE DESCRIPTION	13
3.1 LOCATION	13
3.2 NATURAL ENVIRONMENT.	13
3.3 NATURAL WETLANDS	15
<u>3.3.1 Definition and Classification.</u>	15
<u>3.3.2 Classification of the Swamp Gulch Mine</u> <u>Wetland Study Site</u>	16
4.0 SURFACE HYDROLOGY OF A NATURAL WETLAND	18
4.1 METHODS.	18
4.2 RESULTS AND DISCUSSION	19
5.0 HYDROGEOLOGY OF A NATURAL WETLAND.	24
5.1 LITERATURE REVIEW.	24

	<u>PAGE</u>
5.1.1	<u>Hydraulic Principles of Tracer Use.</u> 24
5.1.2	<u>Tracer Flow Retardation and Tracer Dilution</u> 26
5.1.3	<u>Tracer Methods.</u> 28
5.1.4	<u>Tracer Problems</u> 29
5.1.5	<u>Bromide as a Tracer</u> 30
5.1.6	<u>Wetland Hydrology</u> 34
5.2	MATERIALS AND METHODS 38
5.2.1	<u>Site Instrumentation.</u> 38
5.2.2	<u>Aquifer Characteristics</u> 38
5.2.3	<u>Hydraulic Conductivity.</u> 38
5.2.4	<u>Tracer Experiment</u> 41
5.3	RESULTS AND DISCUSSION. 42
5.3.1	<u>Aquifer Characteristics</u> 42
5.3.2	<u>Hydraulic Conductivity.</u> 47
5.3.3	<u>Tracer Experiment</u> 52
5.4	SUMMARY 63
6.0	WETLAND INFLUENCE ON MINE DRAINAGE WATER QUALITY 65
6.1	SAMPLE COLLECTION AND PROCESSING PROCEDURES 65
6.2	ANALYSES METHODS. 67
6.3	QUALITY CONTROL/QUALITY ASSURANCE 67
6.3.1	<u>Accuracy.</u> 67
6.3.2	<u>Precision</u> 70
6.3.3	<u>Cross Contamination</u> 70
6.4	SURFACE WATER QUALITY 70
6.4.1	<u>Swamp Gulch Above the Carbonate Mine.</u> 70
6.4.2	<u>Swamp Gulch Below the Carbonate Mine.</u> 72
6.4.3	<u>Blackfoot River</u> 74
6.4.4	<u>Background Statiscal Comparisons.</u> 74
6.4.5	<u>Wetland Surface Water</u> 75
6.5	GROUND WATER QUALITY. 76
6.5.1	<u>Shallow Ground-water System</u> 76
6.5.2	<u>Deep Ground-water System.</u> 82
6.5.3	<u>Acid Mine Drainage Seepage Wells.</u> 85
6.6	SUMMARY 85
7.0	DISSOLVED METAL LOADING OF A NATURAL WETLAND. 87
7.1	METHODS 87
7.2	RESULTS AND DISCUSSION. 88
8.0	WETLAND GEOCHEMISTRY AND THE CONTROL OF ACID MINE DRAINAGE. 89
8.1	METHODS 89
8.2	RESULTS AND DISCUSSION. 90
8.2.1	<u>Areal Distribution of Selected Elements</u> 90
8.2.1.1	<u>Iron</u> 90
8.2.1.2	<u>Manganese.</u> 92
8.2.1.3	<u>Aluminum</u> 93
8.2.1.4	<u>Copper</u> 94

	<u>PAGE</u>
8.2.1.5 Lead	94
8.2.1.6 Zinc	97
8.2.1.7 Arsenic.	98
8.2.1.8 Barium and Beryllium	98
8.2.1.9 Cadmium.	98
8.2.1.10 Cation Exchange Capacity	99
8.2.2 Distribution of Elements with Depth	99
8.3 SUMMARY	101
9.0 WETLAND VEGETATION AND THE CONTROL OF ACID MINE DRAINAGE. . .	103
9.1 SAMPLE COLLECTION, PREPARATION AND ANALYSIS	103
9.2 DATA QUALITY ASSURANCE/QUALITY CONTROL.	104
9.3 RESULTS AND DISCUSSION.	105
9.3.1 Vegetation Patterns	105
9.3.2 Canopy Coverage and Production.	105
9.3.3 Element Concentrations in Plant Material.	108
9.3.3.1 Effects of Carex rostrata in Remediating AMD.	110
9.3.3.2 Effects of Salix boothii in Remediating AMD.	115
9.3.3.3 Effects of Betula glandulosa in Remediating AMD.	118
9.3.3.4 Effects of Bryophytes in Remediating AMD.	118
9.3.4 Summary of Element Enrichment in Plants	121
9.3.5 Vegetation Element Levels and Wildlife Hazards	123
9.3.5.1 Dietary Hazard Levels for Animals.	123
9.3.5.2 Wildlife Observed and at Risk.	124
9.3.5.3 Hazards to Wildlife at the Swamp Gulch Wetland.	125
10.0 MICROORGANISM AND THE IMMOBILIZATION OF HEAVY METALS IN WETLANDS.	126
10.1 BACKGROUND.	126
10.2 PHASE I: SWAMP GULCH WETLAND BUFFERING CAPACITY AND THE EFFECT OF SOIL STERILIZATION ON WETLAND PH.	127
10.2.1 Introduction	127
10.2.2 Technical Approach	127
10.2.3 Materials and Methods.	127
10.2.3.1 Soil Sampling.	127
10.2.3.2 Titration Experiment	127
10.2.3.3 Microbiological Experiment	128
10.2.4 Results and Discussion	129
10.2.4.1 Titration Experiment	129
10.2.4.2 Microbiological Experiment	129
10.3 PHASE II: COMPARISON OF NATURAL AND CONSTRUCTED WETLAND SOILS IN THE REMOVAL OF ACIDITY, SULFATE AND IRON.	131

	<u>PAGE</u>
<u>10.3.1 Introduction</u>	131
<u>10.3.2 Technical Approach</u>	131
<u>10.3.3 Materials and Methods</u>	132
<u>10.3.3.1 Field sampling</u>	132
<u>10.3.3.2 Soil core flow characteristics</u>	135
<u>10.3.3.3 Sampling and preparation of AMD.</u>	135
<u>10.3.3.4 Sulfate and iron determinations.</u>	135
<u>10.3.4 Results and Discussion</u>	136
<u>10.3.4.1 Acidity determinations</u>	136
<u>10.3.4.2 Sulfate determinations</u>	140
<u>10.3.4.3 Iron determinations.</u>	145
 10.4 PHASE III: MICROBIAL BIOMASS, COMMUNITY STRUCTURE AND PHYSIOLOGICAL STATUS ASSESSMENT OF NATURAL AND CONSTRUCTED WETLANDS.	 149
<u>10.4.1 Introduction</u>	149
<u>10.4.2 Technical Approach</u>	149
<u>10.4.3 Materials and Methods.</u>	150
<u>10.4.3.1 Field sampling</u>	150
<u>10.4.3.2 Statistical analyses</u>	150
<u>10.4.3.3 Lipid nomenclature</u>	150
<u>10.4.4 Results.</u>	151
<u>10.4.4.1 Total Biomass.</u>	151
<u>10.4.4.2 Community structure.</u>	151
<u>10.4.4.3 Physiological stress</u>	156
<u>10.4.5 Discussion</u>	159
 11.0 LITERATURE CITED.	 162
 APPENDICES.	 178
APPENDIX A: Analytical Methods and Quality Assurance/Quality Control Statistics.	 178
APPENDIX B: Influent Flow Rates, Precipitation and Water Chemistry Data from the Swamp Gulch Wetland	 187
APPENDIX C: Chemical and Physical Data from Swamp Gulch Wetland Sediments	 199
APPENDIX D: Precision and Accuracy of Vegetation Data, and Plant Plant Species Observed at the Swamp Gulch Wetland	 210

LIST OF TABLES

<u>TABLE</u>		<u>PAGE</u>
4-1	Beaver dam tree ring counts.	21
5-1	Range of important physical characteristics of fibric, hemic, and sapric plant materials from northern Minnesota bogs.	37
5-2	Wetland stratigraphy and well completion data.	45
5-3	Field hydraulic conductivity (K), effective porosity (ne), average hydraulic gradient (dh/dl) and calculated average velocity (V) for each site.	51
5-4	Average hydraulic conductivity (K) from field results and calculated average velocity (V).	52
5-5	Bromide background concentrations in the tracer study area of the Swamp Gulch Mine Wetland.	53
5-6	Tracer velocities determined for each site, time and distance from tracer inputs and peak concentrations of bromide.	60
5-7	Tracer velocity comparisons.	61
6-1	Accuracy results based on blind field standards.	68
6-2	Accuracy results based on laboratory spikes.	69
6-3	Precision, based on laboratory duplicates, at the 90% confidence level.	71
6-4	Precision, based on field replicates, at the 90 confidence level.	72
6-5	Field blank.	73
7-1	Total dissolved metal loading on wetland, April 1987 through March 1988.	88
8-1	Mean selected elemental concentration changes with depth for background and AMD impacted areas.	100
8-2	Acrotelm metal masses and the relative efficiency of wetland metal removal.	101
9-1	Mean percent canopy coverage and percent frequency for plant species.	107

TABLEPAGE

9-2	Peak standing crop (kg/ha) at the Swamp Gulch site and at the Hardscrabble background site.	108
9-3	Concentration (ug/s) of elements in <u>Carex rostrata</u>	109
9-4	Water quality in Swamp Gulch above and below the Carbonate mine site, and below the Swamp Gulch wetland at station F-1.	110
9-5	Range and mean element concentrations (ug/g) in aquatic forbs and grasses (from Hutchinson 1975).	111
9-6	Concentration (ug/g) of elements in <u>Salix boothii</u>	116
9-7	Concentration (ug/g) of elements in <u>Betula glandulosa</u>	119
9-8	Accumulation of elements by bryophytes in various environments.	120
9-9	Concentration (ug/g) of elements in bryophytes.	122
9-10	Plant species demonstrating element enrichment (i.e. greater than background levels) at the Swamp Gulch wetland site.	123
9-11	Maximum tolerable levels of dietary minerals for domestic animals.	124
10-1	pH and iron values for AMD incubated under various treatment conditions after a period of nine days.	130
10-2	Swamp Gulch sulfate concentrations vs. flow rate.	144
10-3	Sand Coulee sulfate concentrations vs. flow rate.	144
10-4	Sand Coulee iron concentration vs. total flow volume eluted at end of study.	149
10-5	Fatty Acid (FA) mole % for the various sampling sites.	154
10-6	Description of phospholipid ester-linked fatty acids (PLFA) used to define microbial groups for data interpretation.	155
10-7	Significant difference map generated from Tukey's HSD test.	157

LIST OF FIGURES

<u>FIGURE</u>	<u>PAGE</u>
3-1 Location of Swamp Gulch Wetland Study site in Montana. . . .	14
4-1 Swamp Gulch Wetland Study site (with water sampling stations).	19
4-2 Topographic map with 0.5 m contours.	20
5-1 Breakthrough curves of four different tracer types.	25
5-2 One dimensional example of movement by molecular diffusion.	26
5-3 One dimensional example of hydrodynamic dispersion for tracer particles A-G.	27
5-4 Ranges of published field data on K (hydraulic conductivity) of peat.	34
5-5 Example of study site instrumentation and lithology.	39
5-6 Map of auger hole grid.	40
5-7 Acrotelm isopach map with 0.2m (0.7 ft) contours.	43
5-8 Catotelm isopach map with 0.5 m (1.6 ft) contours.	44
5-9 Deep piezometer map on 7/27/87 with 0.5 m contours.	48
5-10 Shallow piezometer map on 7/27/87 with 0.5 m contours.	49
5-11 Potentiometric surface map on 8/14/87 with 0.025 m contours.	50
5-12 Typical calibration curve generated for the tracer study.	55
5-13 Tracer study breakthrough curve for site AH-14, 6.1 m from tracer input showing little effect of sorption.	56
5-14 Example of infrequent sampling effect on breakthrough curve for site AH-9, 6.1 m from tracer input.	57
5-15 Breakthrough curve representing two tracer plumes at site AH-19, 9.1 m from tracer input.	58
5-16 Steep breakthrough curve indicating adequate tracer mixing at site AH-3, 3 m from tracer input.	58
5-17 Bromide concentration isopach map, 0.95 to 1.5 hours from tracer start (20 mg/L contours).	62

<u>FIGURE</u>	<u>PAGE</u>
6-1 Shallow ground-water types for the wetland system.	77
6-2 Iron concentrations for the wetland shallow ground-water system.	78
6-3 Manganese concentrations for the wetland shallow ground-water system.	79
6-4 Sulfate concentrations for the wetland shallow ground-water system.	80
6-5 Total dissolved solids concentration for the wetland shallow ground-water system.	81
6-6 Iron concentrations for the wetland deep ground-water system.	83
6-7 Manganese concentrations for the wetland deep ground-water system.	84
8-1 Iron concentrations in the wetland acrotelem.	91
8-2 Manganese concentrations in the wetland acrotelem.	92
8-3 Aluminum concentrations in the wetland acrotelem.	93
8-4 Copper concentrations in the wetland acrotelem.	95
8-5 Lead concentrations in the wetland acrotelem.	96
8-6 Zinc concentrations in the wetland acrotelem.	97
9-1 Vegetation associations and canopy coverage transects. . .	106
10-1 Location of wetland sites.	128
10-2 Titration of Swamp Gulch soil with AMD.	129
10-3 Diagram depicting soil core containers used in Phase II greenhouse study.	133
10-4 Sand Coulee constructed wetland sampling site (Heil and Kerins 1988).	134
10-5 Influent (eluent) AMD pH measured over the study period. . .	137
10-6 Swamp Gulch soil core's effluent (eluate) pH measured over study period.	138

<u>FIGURE</u>	<u>PAGE</u>
10-7 Sand Coulee soil core's effluent (eluate) pH measured over study period.	139
10-8 Influent (eluent) AMD iron and sulfate concentrations measured over study period.	141
10-9 Swamp Gulch and Sand Coulee soil core effluent (eluate) sulfate concentrations measured over study period.	143
10-10 Swamp Gulch soil core effluent (eluate) dissolved total and ferrous iron concentrations measured over study period.	146
10-11 Sand Coulee soil core effluent (eluate) dissolved total and ferrous iron concentrations measured over study period.	148
10-12 Total microbial biomass as measured by analysis of phospholipid ester-linked fatty acids.	152
10-13 Sulfate-reducing bacterial phospholipid fatty acid biomarker expressed as density per gram of soil and proportion of total microbial biomass.	153
10-14 Microbial community <u>trans/cis</u> and cyclopropyl/ <u>cis</u> phospholipid ester-linked fatty acids ratios.	158

1.0 INTRODUCTION

1.1 ACID MINE DRAINAGE

Acid mine drainage (AMD), from coal mining in the East and both coal and hard-rock mining in the West, is one of the most persistent environmental pollution problems in the United States. Acid water draining from these sites lowers ground and surface water quality, impacts aquatic and terrestrial biota, and degrades domestic water supplies. An estimated 11,000 miles of rivers and streams in Appalachia have been negatively impacted by AMD (Brodie et al. 1986). More than 5,000 miles of waterways in the East and mid-West fail to meet federal water quality standards owing to acidic mine water (Kim et al. 1982). In Colorado alone, an estimated 10,000 abandoned and inactive metal mines are point sources of AMD, and at least 25 watersheds and 450 stream miles have been rendered barren of aquatic life (Guertin et al. 1985).

Artificial wetlands have been proposed as an approach to provide low-maintenance, broadly applicable solutions to AMD problems and numerous state agencies are currently investing considerable sums of money in the construction of wetlands to treat AMD. The efficacy of these projects is variable (Burris 1984). Some wetlands improve water quality to meet drinking water standards. Others are less effective and provide only temporary metal uptake, selective metal uptake, seasonal effectiveness, or even complete abatement failure. Poor wetland performance may be attributed to inappropriate hydrology or vegetation. Flow channeling may cause polluted discharge to bypass remediation stages; transplanted aquatic plants may be unable to tolerate AMD loading or to take up sufficient metals to significantly abate AMD (McHerron 1985, Wieder and Lang 1986). However, some wetlands with improved hydrologic conditions and well-established metal-tolerant plants still do not perform well, suggesting that other factors, such as microbial activity and sediment geochemistry, are of overriding importance at some sites (Kleinmann and Girts 1986, Lang and Wieder 1985, Peccia & Associates 1987).

The wetlands characterized in this study afforded a unique opportunity to observe the effects of more than 55 years of mining related effluent and AMD upon a natural wetland (Swamp Gulch wetland). In addition, this wetland system could be compared to a non-polluted system, and to a recently constructed AMD wetland treatment system located near Sand Coulee, Montana. The findings will be valuable in the design of future artificial AMD wetland treatment systems by the Montana Abandoned Mine Reclamation Bureau.

1.2 STUDY OBJECTIVES

Specific objectives of this study were to:

- 1) determine the areal and vertical gradients of metal loading in the natural wetland;
- 2) assess contaminant transit time in the natural wetland;
- 3) quantitatively describe plant associations and determine production by dominant species in the natural wetland;
- 4) assess metal loading of dominant vegetation and potential impacts to wildlife in the natural wetland;
- 5) estimate the effective longevity of the wetland to control the acid mine drainage in the natural wetland;
- 6) correlate wetland mass and volume as a function of the quality and quantity of acid mine drainage in the natural wetland; and
- 7) compare the relative biological activity of the natural wetland to the artificially constructed wetland.

1.3 CONCLUSIONS

1.3.1 Wetland Hydrogeochemistry

1. The Swamp Gulch natural wetland is characterized by three basic stratigraphic units: 1) the surficial material (average thickness 0.6 m) composed of undecomposed organic debris, referred to in this report as the acrotelm; 2) the intermediate gray muck zone (average thickness 2.2 m) consisting of decomposed organic materials, silt and clay with occasional sandy gravel lens, generally referred to in this report as the catotelm; and 3) the underlying alluvial gravel zone.
2. Trace metal concentrations were highest in the acrotelm near the Swamp Gulch discharge point and decreased both laterally and vertically away from this point.
3. The only catotelm interval notably impacted by the acid mine drainage was near the discharge point, with trace metal levels in the remaining catotelm being approximately equal to measured background concentrations.

4. Trace element concentration distribution in the underlying alluvial gravel material showed only a nominal association with the Swamp Gulch acid mine discharge.
5. The present iron treatment area of acid mine drainage is confined to approximately 3.9 ha (9.6 acres) and represents an acrotelm volume of about 23,000 m³ with an average thickness of 0.6 m. This volume treats a daily acid mine drainage inflow of 45 m³ containing a volume weighted average of 21 mg/L iron, 3.9 mg/L manganese, 6.5 mg/L aluminum, 2.7 mg/L zinc and 0.71 mg/L copper at a pH of 4.0. The system has in the past, been effective in reducing iron levels in effluent waters.
6. Approximately 550 metric tons of iron have been deposited in the wetland acrotelm from the Swamp Gulch source during the past 55 years. Under present (1987) loading rates (338 kg/year) this represents a 1600 year accumulation, which strongly suggests past loading rates were much higher than those observed during the study period.
7. Conditions conducive to the physio-chemical precipitation of iron may not be conducive to creating optimum conditions for microbial buffering of pH and sulfate reduction. This may explain why Swamp Gulch cores in the laboratory were ineffective in removing iron but, at the same time, the above mentioned iron was present in the acrotelm.
8. Lead is apparently effectively removed from the AMD within the wetland. The AMD attributed acrotelm lead mass is 1.3 metric tons which represents 2,350 years of deposition at present loading rates.
9. The acrotelm copper mass due to AMD is approximately four metric tons which represents 325 years at the observed annual loading rate. This metal is apparently only partially ameliorated within the wetland.
10. Zinc, manganese and cadmium acrotelm masses due to AMD are 6.0, 1.1 and 0.004 metric tons, respectively. These represent deposition intervals of 137, 16.9 and 6.7 years, respectively. It is clear that the wetland efficiency in removal of these metals is limited.
11. Using lead as a basis of comparison of the Swamp Gulch wetland efficiency in removing metals from AMD, the removal rates are: iron, 70 percent; copper, 14 percent; zinc, 5.8 percent; manganese, 0.7 percent; and cadmium, 0.3 percent.

12. Bromide was a good choice for tracing water movement in the natural wetland studied. Employing a natural flow tracer technique and the bromide specific ion electrode method of analysis, sufficient bromide concentrations were detected.
13. The most representative flow velocity of contaminated Swamp Gulch water in the wetland is that determined from a bromide tracer test. The tracer determined mean flow velocity is 2.6×10^{-2} cm/s (73.7 ft/day) and ranged from 3.1×10^{-1} to 6.2×10^{-4} cm/s. These estimated velocities are quite variable and represent a low flow regime (Autumn).
14. Field hydraulic conductivity (K) tests were performed on auger holes and wells in the area of expected metal loading and tracer movement. These hydraulic conductivity estimations ranged from 1.0×10^{-2} to 1.5×10^{-5} cm/s with a mean of 8.3×10^{-4} cm/s. The mean water flow velocity determined from the field hydraulic conductivity values is 7.1×10^{-5} cm/s.
15. The large difference (three orders of magnitude) between mean hydraulic conductivity determined flow velocity and tracer determined flow velocity infers the measurement of two different flow systems. The tracer velocity represents preferential flow paths (channelling) within the shallow flow system (acrotelm) and the hydraulic conductivity determined velocity represents that flow found at greater depth (catotelm).
16. The distance the AMD water travels, before problem amelioration, as estimated by the iron impacted area, is 200m. Using this 200m distance and the average tracer velocity, the average residence time of AMD water in contact with the wetland soils is estimated to be nine days.

1.3.2 Wetland Vegetation/Wildlife

17. The Swamp Gulch wetland was dominated by sedge (Carex rostrata), followed by birch (Betula glandulosa var hallii), lodgepole pine (Pinus contorta), willow (Salix boothii) and moss (Sphagnum tenellum) with percent canopy coverage of 55.5, 11.3, 6.3, 5.5 and 4.7, respectively. Mesic sites were dominated by nearly pure stands of Carex rostrata while the more xeric sites were characterized by Douglas fir (Pseudotsuga menzeisii), Englemann spruce (Picea engelmannii) and subalpine fir (Abies lasiocarpa) and other species typical of the surrounding upland forest.
18. The composition of plant species in the Swamp Gulch wetland was similar to that of the Hardscrabble Creek background wetland.

19. Above-ground production at the Swamp Gulch wetland (3820 kg/ha) was not significantly different than the Hardscrabble Creek background site (3750 kg/ha).
20. Metal levels were generally elevated above background in the tissues of the dominant plant species. Levels were usually higher in below-ground rather than above-ground plant material.
21. Carex rostrata accumulated elements in the following order: Cu>Pb>Al>Fe>Cd>Zn. This species apparently possesses an exclusionary mechanism which prevents the absorption of Mn and Ni.
22. Of the shrubs species studied, Salix boothii accumulated elements to a greater concentration than did Betula glandulosa. Manganese and Ni were readily absorbed by these species as were Cd, Cu, Fe, Pb, Mn, Ni and Zn.
23. The bryophyte Isopterygium pulchellum had tissue metal concentrations that exceeded background levels for Al, Cd, Cu, Fe, Pb, Mn, Ni and Zn.
24. Some metal concentrations exceeded the maximum tolerable dietary intake levels for domestic animals, suggesting a potential problem for animals that consume these plants. The greatest risk was from high levels of Cd.

1.3.3 Wetland Microbiology

25. Removal of acidity in Swamp Gulch natural wetland sediment occurred through the action of anaerobic microorganisms and was not solely a function of abiotic soil characteristics.
26. Based on laboratory and greenhouse experiments, acidity remediation occurred in the Swamp Gulch natural wetland but not in the Sand Coulee constructed wetland.
27. Sulfate-reducing bacteria were present at all experimental sites, and appeared to be reducing sulfate in the Swamp Gulch sediments, but not the Sand Coulee wetland sediments.
28. Adequate carbon and energy levels to support the growth of a variety of indigenous microorganisms including sulfate-reducing bacteria are evident in the Swamp Gulch sediments.
29. In the laboratory experiments, iron removal from AMD was not evident in either the Swamp Gulch or Sand Coulee wetland cores.

30. The total microbial biomass and densities of sulfate-reducing bacteria were low in the Sand Coulee constructed wetland sediments, probably because of insufficient levels of carbon and other nutrients.
31. The microbial populations at both Swamp Gulch and Sand Coulee wetlands were physiologically stressed. This may account for the inability of the Sand Coulee sediments to facilitate acid, iron or sulfate removal. Furthermore, it may be that the Swamp Gulch sediment microbiota are not able to achieve their maximum physiological capabilities for AMD amelioration.
32. It should not be automatically assumed that poor performance in a constructed wetland is due solely to wetland size or hydrology. There is a significant microbiological contribution in working wetlands that should be transferable to non-working wetlands.

1.4 RECOMMENDATIONS

1.4.1 Wetland Hydrogeochemistry

1. These data suggest that any constructed wetland should be made as large as practically possible. It is doubtful that any constructed wetland could be built as large as the impacted area of this natural wetland (3.9 ha).
2. The catotelm (muck zone) was generally ineffective in removing metals from AMD, but a limited thickness of this material (probably less than 7 cm) may be needed for a proper substrate for some types of wetland vegetation.
3. The thickness of the natural wetland acrotelm averages 0.6 m. This thickness may be desirable in constructed wetlands operating under similar circumstances.
4. The specific ion electrode method of bromide tracer analysis could be improved by periodic double sampling and analysis with a different method (titration), especially at low bromide concentrations.
5. The acquisition of limited additional data from the Swamp Gulch wetland site would allow a determination of the system's present efficiency in removing metals from AMD. This will help determine the effective treatment longevity of the wetland.

6. Precipitation of iron by physio-chemical processes may be a very important mechanism for removing iron from AMD (Wieder and Lang 1986) and likely played an important role in the Swamp Gulch wetland. Further evaluation of the wetland iron is recommended to determine if this material is organically derived or inorganic oxide precipitation.

1.4.2 Wetland Vegetation/Wildlife

7. Carex rostrata, Betula glandulosa and Salix boothii should be strongly considered for transplantation to constructed wetlands. These plants absorbed substantial quantities of Al, Cd, Cu, Fe, Pb, Mn, Ni, and Zn without ill effect. These plants are common to many parts of Montana and should be relatively easy to transplant. It is therefore recommended that a screening study be implemented to determine the ability of these species to tolerate transplantation and to proliferate at a constructed wetland.
8. Potential impacts to wildlife from high plant tissue metal levels should be investigated. The concern has been expressed that in constructing wetlands for the amelioration of AMD one is increasing the risk of poisoning the wildlife resource.
9. The best indication of whether animals are at risk at the Swamp Gulch wetland is to analyze tissue samples from those animals that are most likely to be poisoned. It is therefore recommended that voles (Microtus spp.) be trapped and their kidneys be analyzed for metals, especially Cd.
10. Until it can be demonstrated that animals will not be poisoned by consuming vegetation at constructed AMD wetlands, it is recommended that these systems be fenced to limit wildlife access.

1.4.3 Wetland Microbiology

11. Procedures need to be established to transfer microbiological activity of a working wetland to an ineffective wetland.
12. The ester-linked phospholipid fatty-acid technique is an important tool for assessing the impacts of AMD on indigenous microbiota and in evaluating the efficacy of constructed wetlands.
13. The addition of carbon and other nutrient sources to wetlands would diversify the microbial community structure and be especially beneficial to the sulfate-reducing bacterial populations.

14. Wetland construction practices should provide as much cation exchange capacity as possible to aid biological removal acidity and metals. Acidity control through the application of appropriate amendments should also be considered.
15. Wetland construction practices should provide favorable conditions, such as anaerobiosis (e.g., by varying constructed wetland depth), for the proliferation of sulfate-reducing bacteria. If these bacteria are able to reduce sulfate to sulfide, metals should be immobilized through the formation of metal sulfide precipitates.
16. Seasonal effectiveness of wetlands needs to be assessed in the context of the microbiota and site geochemistries.
17. Emphasis on the exact nature of physiological stresses imposed on microbial populations in wetlands need to be elucidated.
18. Vertical and lateral zones of microbial activity need to be mapped in both natural and constructed wetlands in order to develop engineering criteria for wetland construction.

2.0 NATURAL SYSTEMS FOR AMD CONTROL

2.1 WETLAND TREATMENT SYSTEMS

Research has confirmed the theory that natural wetlands can improve the quality of acid mine water flowing through them (Wieder and Lang 1984, 1986). However, since natural wetlands are seldom located adjacent to acid mine effluent and because of legal barriers regarding the discharge of pollutants into wetlands, they are seldom utilized for AMD abatement. Constructing artificial wetlands is the obvious alternative to using natural wetlands.

Artificial systems, both in the laboratory and in the field, have demonstrated the ability to improve water quality by removing certain metal ions (Kleinmann et al, 1983, Burris et al. 1984, Gerber et al. 1985, Wieder et al. 1985 and Brodie 1986). These systems utilize bryophytes, particularly Sphagnum spp., which have generally been successful in removing Fe, Mg, Mn and other metals from solution. Simola (1977) and Chaney and Hundemann (1979) studied the ability of Sphagnum to absorb and tolerate Cd. Bryophyte species of the genera Rhynchostegium, Rhytidiadelphus, Amblystegium and Fontinalis have demonstrated metallophytic properties (Wehr and Whitten 1983, Brown and Beckett 1985). Emergent vascular plant species (Typha, Carex, Eleocharis and Juncus) and floating plants (Eichhornia, Salvinia and Lemna) frequently occur in areas with acidic water and/or high metal concentrations. Haag and Covert (1987) reported Typha angustifolia and several grass species growing in AMD water in Missouri. Dinges (1982) reviews natural and artificial systems for improvement of polluted waters. Species of Typha are able to tolerate high metal levels, and therefore have been extensively studied (Taylor and Crowder 1983a, 1983b, 1984, McNaughton et al. 1974, Mayer and Gorham 1951, Boyd 1970 and Bayly and O'Neill 1972). In terms of biofiltration of AMD, vascular plants have not been studied as intensively as bryophytes. The ability of vascular plants to thrive in AMD water indicates that more research is needed in this area.

Mechanisms thought to operate in pollution control include cation exchange of metal ions on the surface of organic matter, physical filtration, chelation or complexation by organic compounds, dilution, and biological sorption by bacteria, algae, fungi, animals and plants. Guertin et al. (1985) presents a review of each of these mechanisms.

Field testing of artificial wetlands are underway at over 20 sites in the eastern United States (Grits and Kleinmann 1986), with the possibility of hundreds of wetlands being constructed in

Appalachia in the next few years. Guertin et al. (1985) discuss a wetland system installed in Colorado and Heil and Kerins (1988) report results from an artificial wetland in central Montana.

Primary design criteria for construction of artificial wetlands are based on estimates of system loading rates, since these directly effect maximum treatment efficiency and capacity of the wetland. Influent volume and chemical composition in conjunction with treatment rate and required effluent concentrations provide the basis for determining the optimal AMD to wetland volume ratio, flow path length, and detention time. Baseline data on flow rate and chemical composition need to reflect yearly extremes. For a review of construction parameters and essential baseline information, consult Girts and Kleinmann (1986), Pesavento (1984) and Brooks (1984). Girts and Kleinmann (1986) present important criteria used for selecting, planting and transplanting various plant species.

2.2 BIOSORPTION OF ELEMENTS BY PLANTS

The idea that plant species absorb nutrients from the soil was first suggested by Glauber in 1656. Prior to this, it was generally accepted that plants acquired all their sustenance from air and water. In 1804, de Saussure demonstrated that plants acquire a portion of their constituents from the medium in which they are rooted. Hewitt (1975) reported that higher plants absorb a portion of all the elements that are present in the rhizosphere solution, but exhibit absorption selectivity.

Vascular plants characteristic of soils with high metal concentrations have been known for centuries. In 1588, Thalius noted that Minuartia verna was an indicator of metal presence in soil (Ernst 1965). Metallophytes (plants that accumulate metal ions) often have tissue concentrations of ions several times higher than the medium in which they are growing. These types of plants have been used in geologic prospecting to indicate ore bodies (Cannon 1960).

Some bryophytes have also been identified as metallophytes, accumulating elements well in excess of their structural or metabolic requirements. The high ion exchange capacity, simple cellular morphology, and compact low growth form of many mosses can result in the rapid accumulation of metal ions, either by extracellular ion exchange or by entrapment of particulates. Lee et al. (1983) studied the brilliant lime green mosses that flourished in hillsides blanket bogs fed by spring water enriched with Zn, Pb and Cu. Bryophytes belonging to the Bryaceae and Bartramiaceae families were reported useful in identifying Cu and U deposits (Whitehead and Brooks 1969).

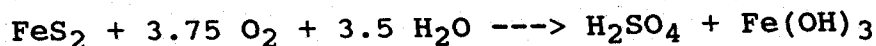
2.3 METALLOPHYTES AND LAND/WATER RECLAMATION

Metallophytes have proven useful in the reclamation of metalliferous waste sites (Gadgil 1968, Johnson et al. 1977), and in the treatment of wastewater (Dinges 1982). The mechanisms of wastewater treatment by natural systems is discussed in detail by Dinges (1982). The use of metallophytes to improve the quality of acid mine water is being studied extensively throughout the United States.

2.4 CHEMICAL PROCESSES OF ACID MINE DRAINAGE

2.4.1 Acid Water Formation

Acid water occurs when pyrite (FeS_2) is oxidized in air and water to form sulfuric acid (H_2SO_4) and iron hydroxide ($\text{Fe}(\text{OH})_3$). Jaynes et al. (1984) and Guertin et al. (1985) explain the stoichiometry of acid formation in detail. The overall chemical reaction was summarized by Schumate et al. (1971) as:



Acidic water is often indicated by the presence of a white to reddish iron hydroxide precipitate known as 'yellowboy'.

2.4.2 Bacterial Mediation

The kinetics of acid formation are dependent on the availability of oxygen, the surface area of the pyrite, the pH of the influent water, and the activity of iron-oxidizing bacteria. The principal bacteria involved in accelerating pyrite oxidation are the chemoautotrophic Thiobacillus thiooxidans (Kleinmann and Crerar 1979). Since these bacteria can increase the rate of this reaction by a factor of over 100 times, they can play a crucial role in acid production. Singer and Strum (1970) concluded that this bacterially mediated reaction was the rate determining step in acid formation. Walsh and Mitchell (1972) believe that the production of acid can be controlled by breaking the Fe-bacteria succession in the pH range of 4.0 to 4.5. Bacteria of the genus Metallogenium have been found to catalyze the acid producing reaction (Parizik 1985), while other microorganisms such as T. thiooxidans are important in bringing iron into solution from soil and organic matter (Oborn and Hem 1962).

2.4.3 Influence of pH

In general, metal ions increase in solubility several orders of magnitude with each unit decrease in pH. The solubility and stability of metals in solution is also a function of Eh, or oxidation-reduction potential. Hem and Cropper (1959) present data on the relationship between Fe ions, pH and Eh. In terms of

AMD treatment systems, Eh-pH diagrams are useful for obtaining rough estimates of the dominant ions in the effluent (Guertin et al. 1985)

Traditional methods for treating acidic water are to oxygenate the water to remove volatile acids such as carbonic acid. Bases such as sodium hydroxide, limestone or sodium carbonate are often used as neutralizing agents. Injecting alkaline materials into acid producing spoil material to prevent AMD from occurring was reviewed by Ladwig et al. (1985).

2.4.4 Pyrite Oxidation Control

Methods for controlling pyrite oxidation include the use of bactericides (Barnes and Romberger 1968, Shearer et al. 1970), the use of bromo-substituted phenols to inhibit metabolism (Aleems 1972), the introduction of heterotrophic bacteria to prey on the iron and sulfur oxidizing bacteria, removal of attachments for the stalked bacterium Metallogenium, and increase the pH in the neighborhood of the bacteria (Walsh and Mitchell 1972). Some of these techniques have been tested in the laboratory while others have merely been suggested.

Anionic surfactants, primarily sodium dodecyl sulfate (SDS), alters the cytoplasmic membrane of the bacteria, allowing hydrogen ions access to the cell. Two organic compounds, sodium benzoate and potassium sorbate, have been demonstrated as effective as SDS (Erickson et al. 1985). Pelletized bactericide has raised the pH of spoil material, allowing it to be revegetated (Zaburnov 1987).

3.0 SITE DESCRIPTION

3.1 LOCATION

The Swamp Gulch Wetland study site is located 14 miles (22.4 km) east of the town of Lincoln, Montana in Lewis and Clark County (section 20, T1N, R6W) (Figure 3-1). The site is bordered on the north by Montana highway 200 and on the south by the Blackfoot River. The entire wetland covers approximately 72.9 ha (180 acres); however, the study area proper covers about two thirds of that area of which 3.9 ha (9.6 acres) is apparently impacted by AMD.

3.2 NATURAL ENVIRONMENT

The wetland study site is in the headwaters of the Blackfoot River at an elevation of about 1573 m (5160 ft). Elevations in the Swamp Gulch drainage which feeds a portion of the study area range from about 1574 m (5165 ft) to 1908 m (6260 ft). The Swamp Gulch watershed contains 72.6 ha (179 acres) and is composed of generally well vegetated southeast to southwest aspects with a total length of about 1.6 km (1 mi). The site is on the west side of the continental divide and is therefore influenced more by Pacific weather patterns than by the continental climate in the region east of the divide. However, moisture from the Gulf of Mexico occasionally spills over the divide during the summer producing intense thunderstorms. Mean annual precipitation is approximately 73.7 cm (29 in) (SCS 1981) much of which occurs as snow (Coffin and Wilke 1971). Precipitation during the 1987 water year and annual year averaged 60 percent of normal at the Lincoln Ranger Station. Record (1949-1987) maximum and minimum annual precipitation at the Lincoln Ranger Station are 91.49 cm (36.02 in) and 31.14 cm (12.26 in), which occurred in 1975 and 1973, respectively. Average January and July temperatures are -7.2°C (19.1°F) and 16.4°C (61.6°F), respectively (NOAA 1987).

The Blackfoot River drainage was significantly influenced by glaciation. Sediments deposited by retreating valley glaciers created a series of moraines and outwash plains in the Lincoln area, allowing for the formation of wetlands in drainage impeded areas on the upslope sides of the moraines (Alt and Hyndman 1972). The lithology of the study area is dominated by the Spokane Shale Formation, part of the Belt series super group. This formation consists of red or red-purple shale with numerous green beds locally with quartzite (Ross et al. 1955). Parent material of the valley floor apparently consists of a thin layer of glacial till with fluvial gravel deposits.

Swamp Gulch and many other drainages of the region have been impacted by mining. The Carbonate Mine is within the Heddleston mining district and in 1933 included an adit about 850 feet long and a shaft approximately 300 feet deep (Pardee and Schrader 1933). A 75 ton mill was erected in the summer of 1944 and it is reported that 2500 tons of lead-zinc ore was taken out in 1945 (Shea 1947).

The geology of the Carbonate Mine is similar to that of the Mike Horse Mine which is the largest mine in the district (Shea 1947). The mine workings are in a diorite sill that intruded argillite of the Spokane Formation, Belt Precambrian sedimentary rocks (McClernan 1983). Intruding into the sill and Belt rocks is a decomposed rhyolite dike (Shea 1947). Pardee and Schrader (1933) describe the lode as consisting of a sheared and altered diorite partly replaced by sulphides. The mine lode contains abundant pyrite in several forms including fine grained, coarse grained and radial. Ore minerals are galena, sphalerite, silver and gold. Gangue minerals occur primarily as sericite, pyrite and quartz and also include "a carbonate containing calcium, iron and a little manganese" (Pardee and Schrader 1933).

The dominant soil in the study area is a Hydric Borofibrist (Soil Survey Staff 1975). The gravels most often encountered at depth are fluvial in origin as evidenced by many subrounded pebbles. These were deposited by the meandering river following glaciation and were likely derived from glacial outwash materials.

Vegetation within the Swamp Gulch wetland is dominated by sedge (Carex rostrata). This species is sometimes associated with shrubs (Betula glandulosa and Salix boothii) or lodgepole pine (Pinus contorta), but is most often encountered in nearly pure stands. Where the water table is relatively low, plant species more indicative of the surrounding forest are found.

Upland forest vegetation is predominantly subalpine fir (Abies lasiocarpa), Douglas-fir (Pseudotsuga menzeisii) and ponderosa pine (Pinus ponderosa) climax forest (Ross and Hunter 1976). Understory species include pinegrass (Calamagrostis rubescens), huckleberry (Vaccinium spp.), bluebunch wheatgrass (Agropyron spicatum), mountain brome (Bromus marginatus) and columbia needlegrass (Stipa columbiana).

3.3 NATURAL WETLANDS

3.3.1 Definition and Classification

Marshes, swamps and bogs have been well known terms for centuries, but only recently have attempts been made to group these ecological systems under the single term "wetland". This

change has developed out of a need to understand and describe the characteristics and values of all types of land so that they can be wisely and effectively managed (Cowardin et al. 1979).

In general, wetlands are lands where the soil or substrate is at least periodically saturated with or covered with water. Saturation with water is the dominant factor in soil development and in determining the types of plants and animals that are in the soil and on its surface.

The most widely used wetland classification system in the United States is that of Martin et al. (1953), which was republished in the U.S. Fish and Wildlife Service Circular 39 (Shaw and Frederick 1956). Wetland types were based on criteria such as water depth, water permanence, water chemistry, vegetation life form and dominant plant species. The system defined by Golet and Larson (1974) refined freshwater wetland types of Circular 39 for the glaciated Northeast, but did not recognize the coastal (tidal) freshwater wetlands. Stewart and Kantrud (1971) devised a system for classifying wetlands in the glaciated prairies. This system also relied on water and vegetation characteristics.

In 1979, a wetland classification scheme was developed as a tool for managers to inventory wetlands and deep water habitats of the United States (Cowardin et al. 1979). This system defines wetlands by plants (hydrophytes), soils (hydric soils) and frequency of flooding. The structure of the system is hierarchical, progressing from system to class, subclass and dominance types. Modifiers for water regime, water chemistry and soils are applied at the class, subclass and dominance levels. Special modifiers are available for man or beaver modified wetlands. This scheme was used to classify the Swamp Gulch study area.

3.3.2 Classification of the Swamp Gulch Wetland Study Site

Based on the Cowardin et al. (1979) scheme, the Swamp Gulch site can be classified as a "Palustrine" system. The Palustrine system includes all nontidal wetlands dominated by shrubs, trees, perennial emergents, or emergent mosses or lichens. This category was developed to group the vegetated wetlands traditionally called marshes, swamps, bogs, fens, prairies or ponds. Palustrine systems may be situated almost anywhere on the landscape.

In terms of the subclass, the majority of the Swamp Gulch site was classified as a "Persistent Emergent Wetland" because of the dominance by erect, rooted and persistent herbaceous hydrophytes. This subclass is typified by such plant types as rush (Juncus), cattails (Typha), sedge (Carex), bulrush (Scirpus) and certain species of true grasses. The portion of the Swamp Gulch site dominated by shrubs was classified as a "Palustrine

Broad-leaved Deciduous Scrub - Shrub Wetland". This subclass is typified by alder (Alnus), willow (Salix), and birch (Betula). The western one-third of the site was classified as "Palustrine Needle-leaved Evergreen Forested Wetland". This subclass is reserved for wetlands having trees taller than 6 meters.

To completely describe the Swamp Gulch site (and any other wetland) certain modifiers at the class level need to be applied. These modifiers are for water regime, water chemistry (Eh and pH), soil factors, and special modifiers for disturbed wetlands. For simplicity, we will limit our discussion of special modifiers to the dominant subclass present at the Swamp Gulch site, the 'Palustrine Persistent Emergent Wetland'. The water regime for this subclass was 'nontidal, semipermanently or seasonably flooded'. These descriptors cover the range from saturated surface soil to areas consisting of open water during the growing season. The salinity modifier was 'fresh' (i.e. less than 800 umhos/cm), the pH modifier was 'acid' (i.e. less than 5.5), and the soil modifier was 'organic'. The special modifier was 'impounded', due to the activities of beaver.

The full wetland classification for the majority of the Swamp Gulch site according to Cowardin et al. (1979) is, "Palustrine, Persistent, Emergent Wetland, nontidal, semipermanent or seasonably flooded, fresh, acid, organic, impounded".

4.0 SURFACE HYDROLOGY OF A NATURAL WETLAND

The Swamp Gulch surface wetland water hydrology was evaluated: 1) to determine the amount of AMD inflow and hence the calculated metal loading input to the wetland (see Section 7.0); 2) to evaluate the general surface flow pattern to determine the area of impact and if channeling was evident; and 3) to evaluate the effect of AMD dilution by other unpolluted water sources flowing into the wetland. The following sections present these aspects of the study.

4.1 METHODS

The measurement of the AMD into the wetland was determined using a 22.9 cm (0.75 ft) throat Parshall type flume equipped with a Stevens Type F recorder. This unit was installed June 15, 1987 in the upstream toe of the tailings dam on Swamp Gulch. The area beneath the flume was saturated and a mixture of tailings and bentonite was used to seal the area. The flume discharged directly into a wooden box culvert through the dam. The box culvert had sufficient cross-sectional area and slope so that no submersion of the flume outlet was observed. Complete records were obtained from June 15 through November 17, 1987, at which time the unit froze. The rate of flow was calculated using the equation:

$$Q = 4WHa^{1.522} W^{0.026} \text{ (Parshall 1950)}$$

where: Q = discharge, cubic feet per second (cfs)
 W = throat width (feet)
 Ha = upper gage head (feet)

The rate calculations were integrated over time intervals varying from 0.25 to 24 hours as determined from the recorder charts for total daily flow.

Precipitation was measured with a Belfort #5-780-6 universal recording rain gage installed near piezometer site E-1 on June 2-3, 1987 (Figure 4-1). The gage was mounted on 3.05 m (10 ft) long by 1.9 cm (0.75 in) inside diameter flanged pipes driven into the wetland approximately 2 m (7 ft). The installation is equipped with an Alter (1937) type wind screen mounted at the orifice level, which is approximately 2.1 m (7 ft) above the wetland surface. The gage was calibrated in 2.54 cm (1 in) increments from 0 to 15.24 cm (0-6 in) following installation. Records are complete to date (Appendix B-2), but some snowfall may have been missed from approximately December 15, 1987 through January 11, 1988, due to snow bridging.

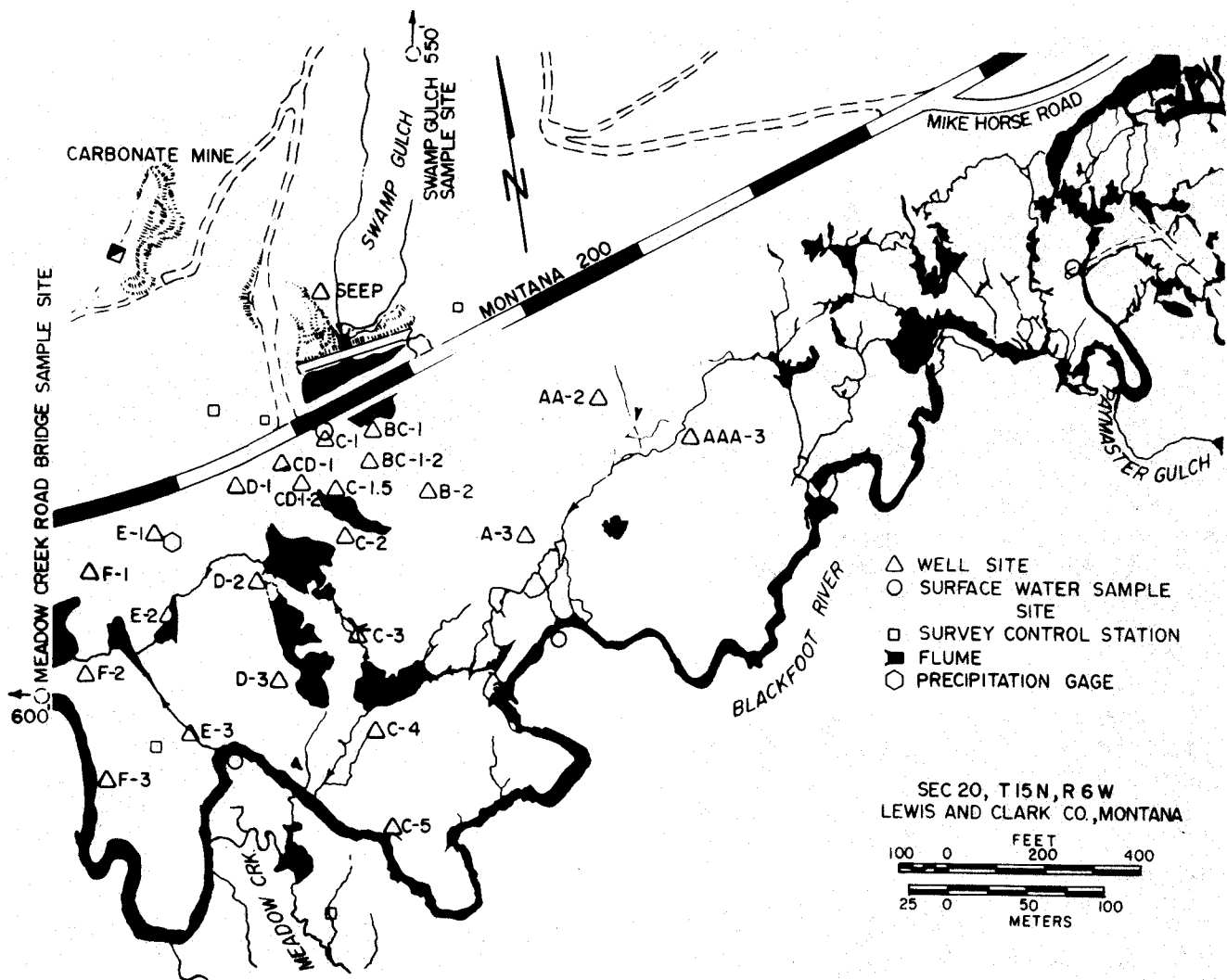


Figure 4-1. Swamp Gulch Wetland Study site (with water sampling stations).

4.2 RESULTS AND DISCUSSION

Inflow of surface water to the wetland site is from two general sources: Swamp Gulch and the Blackfoot River. The Blackfoot River water is dispersed by several beaver dam systems below the confluence with Pass Creek (Figure 4-1). Direct drainage from these dams flows in moderately well-defined channels southwestward past site AAA-3 into the study area. The Blackfoot River water drains into the beaver pond between sites C-3 and C-4 and then, via a well defined channel, flows past site C-3 to the pond directly east of site D-2. The flow at site C-3 may approach 1 cfs at times and may average better than 0.25 cfs. Outflow from the pond east of site D-2 drains westward to a small pond adjacent to site E-2 and then through two ponds north of

site F-2. The pond immediately northeast of F-2 receives additional Blackfoot River water diverted via a channel near site E-3. Drainage from the pond northwest of F-2 flows approximately 37 m (120 ft) in a well defined channel into the Blackfoot River. This system likely determines the current southern extent of the Swamp Gulch acid drainage impact area. The pond northeast of site F-2 is also the likely point of accumulation of wetland drainage resulting from Swamp Gulch. The area topography generally confines this flow to the area immediately south of Highway 200 (Figure 4-2).

It should be noted that fish, Eastern Brook Trout, have been occasionally observed in this system from site AAA-3 to the pond near site E-2. These were observed frequently in the channel

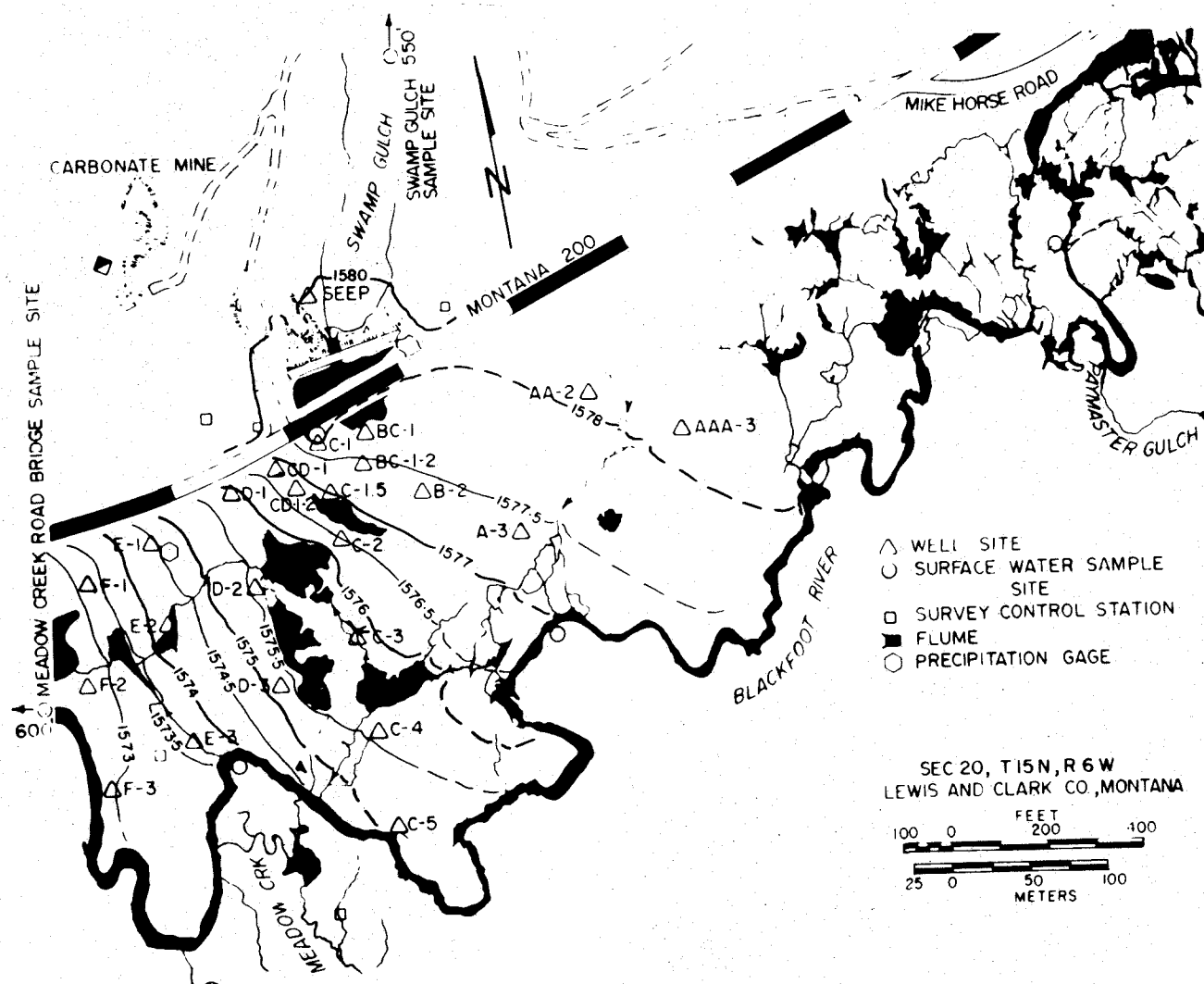


Figure 4-2. Topographic map with 0.5 m contours.

from AAA-3 to the pond between C-3 and C-4, and only once at the pond near E-2 (during the April, 1987 sampling trip).

The area surrounding the Swamp Gulch AMD discharge point is relatively uniform with no distinct channeling. The discharge is spread in a wide arc of roughly 180 degrees.

The age of the present wetland drainage system is important in relation to mining and post-mining periods in determining the actual extent of the area potentially impacted from this source and if there have been any changes in the impacted area. A comparison of aerial photography from the years 1939, 1966, 1972 and 1987 indicates that the present beaver dam system was in place before July 15, 1939, and that apparently mature Pinus contorta were established at that time in the same beaver dam areas where they are presently found. Several dead specimens were obtained and tree ring counts indicated ages from 22 to 70 years (Table 4-1). The oldest specimen exhibited intact bark and some needles suggesting it has been dead less than five years. Other samples had deeply checked surfaces with no bark with tree ring counts ranging from 22-63, which suggests these are some of the dead trees evident in the 1972 photographs. Based on these data, the minimum age for dam systems would be 78 years, having been established sometime prior to 1909. It is likely that the lodgepole pine could not have become well established on these dam areas until unsaturated surface soil conditions developed, possibly following abandonment. It is, therefore, probable that the system has remained essentially unchanged since at least 1900

Table 4-1. Beaver dam tree ring counts.

Sample Number	Ring Counts	Condition	Location
1	70	Bark intact, some needles, 12 inch diameter.	30 ft SW C-3
2	22	No bark, deeply checked, 3.5 inch diameter.	25 ft SW C-3
3	47	No bark, deeply checked, 5.5 inch diameter.	30 ft S D-2
4	27	No bark, deeply checked, 7.5 inch diameter.	25 ft W D-2
5	63	No bark, deeply checked, 8.5 inch diameter.	15 ft N C-4

and the mining impact area has not changed during this interval. The 1939 aerial photographs also indicate that no tailings dam was present adjacent to the highway at that time, which supports the apparent 1944 construction date for this feature.

Flume records indicate flow rates varied from a minimum of 0.1 L/s (0.004 cfs) on November 15, 1987, to a maximum of 14 L/s (0.5 cfs). The peak flow occurred July 17-18, 1987, in response to 6.6 cm (2.60 in) of precipitation received at the site during the two day period. Daily minimum and maximum flows were 12.3m³ (432 ft³) and 734m³ (25,840 ft³), respectively. The average daily flow was 45m³ (1,588 ft³). A comparison of flume hydrographs and precipitation gage charts suggests the time of concentration (the time interval from initial storm precipitation to the storm hydrograph peak) at the flume is about ten hours under conditions of low antecedent precipitation and moderately large storms. The maximum intensity precipitation rate was recorded on July 22, 1987, when 1.4 cm (0.55 in) fell within one hour. The maximum daily total (24 hours) was 5.72 cm (2.25 in) which occurred July 17, 1987 (Appendix B-1). This event was close to the reported ten year, 24 hour storm event of 5.6 to 6.1 cm (2.2 to 2.4 in) (Miller et al. 1973). It is apparent that even the moderately intense storms do not produce peak runoff sufficient to cause any channeling in the wetland. Extreme events would be attenuated by the capacity of the highway culvert and highway overtopping would likely spread the flow to reduce wetland channeling. There is no evidence that this type of event has occurred in the last few years. Normal peak flow would be expected during spring snow melt and no data has yet been recorded for this time interval.

Precipitation records from the site (Appendix B-2) suggest warm season precipitation is, in general, similar to that observed at the Lincoln Ranger Station with the exception of isolated storms such as the July 1987 event. The limited record is insufficient to determine if these isolated storms have a significant effect on the annual long term precipitation. The November 1987 through February 1988 records indicate consistently more snowfall occurs at the wetland than occurs at the Lincoln Ranger Station. This would be expected due to the higher elevation of the wetland. The insufficient record length at the present time prevents an accurate estimate of average annual precipitation, but it is apparent that the site does receive amounts in excess of the 48.1 cm (18.93 in) received at the Lincoln Ranger Station. A linear regression of monthly data from the two sites for the June 1987 through February 1988 records gives the following relationship.

$$\begin{aligned} \text{WP (cm)} &= (1.45) \text{ LRSP} - .47 \text{ cm} \\ \text{where: WP} &= \text{wetland precipitation} \\ \text{LRSP} &= \text{Lincoln Ranger Station precipitation} \\ r^2 &= 0.69 \text{ (r=0.83)} \end{aligned}$$

Using the regression equation with the average annual precipitation at the Lincoln Ranger Station gives a 69.3 cm (27.27 in) estimate of the annual precipitation at the wetland. This is somewhat less than the 73.66 (29 in) estimated from SCS (1981) isolines. A rough free water surface evaporation estimate as determined from Kohler et al. (1959) is approximately 66 cm (26 in).

In summary, the Swamp Gulch AMD daily average flow is estimated to be 45m^3 with a range of 12.3 to 734m^3 . The Swamp Gulch discharge into the wetland is generally well distributed without any notable localized channels, although there are preferred flow paths (see Section 5). Blackfoot River and Pass Creek surface water is introduced into the wetland via beaver dams and two well-defined channels which have apparently been in place during the complete history of the mine. The channels have likely limited the extent of the AMD impact area and provide a sufficient unpolluted water volume to dilute the measured AMD volume to low levels. The Swamp Gulch wetland receives precipitation in excess of that known for the Lincoln Ranger Station. The precipitation is likely also slightly in excess of the estimated free water surface evaporation at the site and hence, no concentration of dissolved solids in wetland water likely occurs from evaporation.

5.0 HYDROGEOLOGY OF A NATURAL WETLAND

The purpose of this part of the Swamp Gulch wetland study was to determine if residence time of mine-contaminated water in a natural wetland could be determined using a bromide tracer. Specific objectives of the study were as follows:

- 1) Investigate the ability of bromide to serve as a tracer in the wetland.
- 2) Identify water flow velocity in the wetland.
- 3) Relate water flow velocity to wetland physical properties (hydraulic conductivity).
- 4) Determine residence time in the wetland area affected by AMD.

5.1 LITERATURE REVIEW

5.1.1 Hydraulic Principles of Tracer Use

In hydrology, a tracer is either matter or energy carried by water which will give information concerning the direction and/or velocity of the water. When sufficient data are collected, tracers can assist in the determination of hydraulic conductivity, porosity, dispersivity and other hydrogeologic parameters (Davis et al. 1980). Introducing a tracer at one point in the flow field and observing its arrival at other points, is the most direct method of determining groundwater velocity (Freeze and Cherry, 1979).

One factor which controls velocity is hydraulic conductivity which represents the ease with which water moves through the soil or aquifer. Hydraulic conductivity, "K", is often used to characterize an aquifer because it includes the properties of the fluid and the field of gravity as well as the properties of the porous medium such as permeability. Hydraulic conductivity has the dimensions of length/time (L/T) or velocity (Fetter 1980). Hydraulic conductivity, porosity and permeability values can vary widely in space and time. The relationship between velocity and hydraulic conductivity is derived from Darcy's Law and is shown in the following equation:

$$V = (-K/ne) (dh/dl) \quad (\text{Eq. 5-1})$$

Where: V = average linear velocity
 K = hydraulic conductivity
 ne = average effective porosity

dh/dl = average hydraulic gradient
 dh = change in head
 dl = change in length

Interpretation of the results of tracer tests involves plotting the concentration of a tracer as a function of time or volume of water passing through the aquifer. In the resulting "breakthrough curve" the concentration is commonly given as a ratio of the measured concentration at the observation well, " C ", to the initial tracer concentration injected, " C_0 ". The average travel time of a non-reactive (conservative) tracer can be determined from a breakthrough curve for transport from the injection point to the observation point. The first arrival time of a tracer as it moves through the system represents the maximum velocity of the groundwater. The peak concentration of the tracer represents the average transit time of groundwater through the system, if a conservative tracer is used. Retardation of the breakthrough curve. Figure 5-1 gives hypothetical examples of breakthrough curves for a mixture of tracers injected as a single slug into an aquifer. Notice the change in the curve shape when the tracer is not conservative and interacts with the aquifer system.

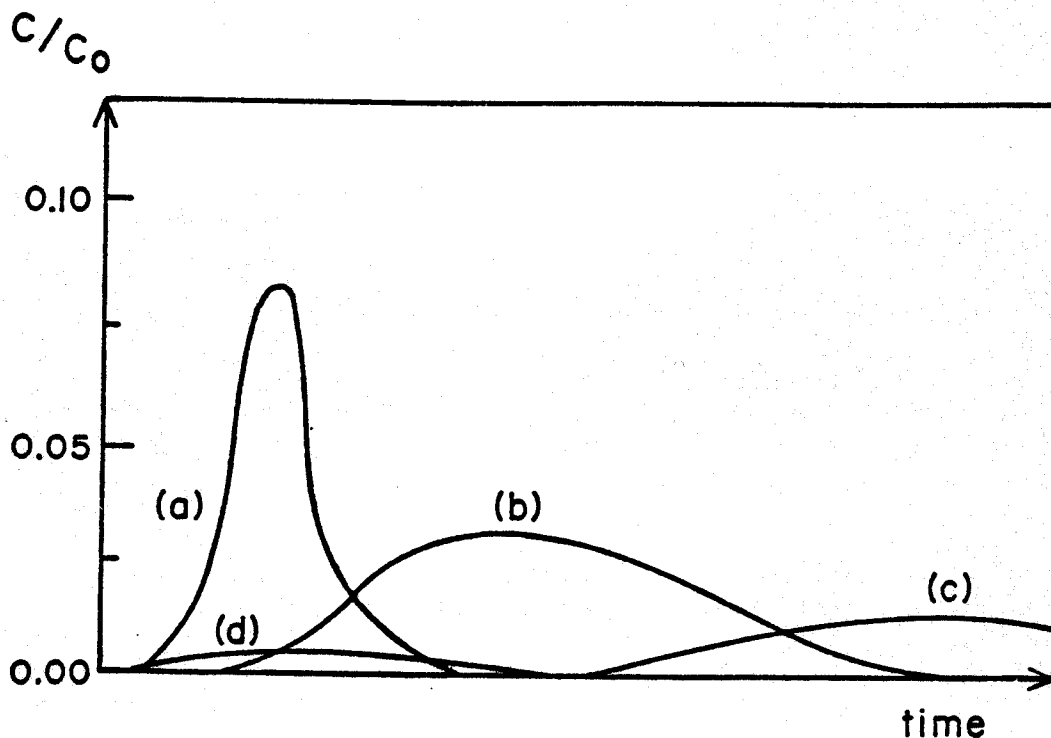


Figure 5-1. Breakthrough curves of four different tracer types: (a) is conservative, (b) some effect of sorption, (c) large effect of sorption, and (d) precipitated or destroyed (From Davis et al, 1985).

5.1.2 Tracer Flow Retardation and Tracer Dilution

Tracers are not perfectly conservative and the concentration distribution of a water soluble substance which is transported in a porous medium by groundwater is affected by sorption, molecular diffusion and hydrodynamic dispersion (Gustafsson and Klockars 1981). Sorption includes adsorption and absorption processes. The many chemical processes which contribute to sorption result in retardation of tracer movement. Thus, velocity of the tracer is slower than that of the groundwater. Therefore in order to design a meaningful tracer experiment, the sorptive characteristics of the tracer must be known (Davis et al. 1985). Davis et al (1985) and Gustafsson and Klockars (1981) give equations to describe the effect of tracer sorption in relation to groundwater flow.

Hydrodynamic dispersion and molecular diffusion have the effect of diluting the concentrations of artificially induced tracers. Hydrodynamic dispersion generally affects short term tracer tests and molecular diffusion affects the concentration of slow moving tracers in heterogeneous materials (Davis et al. 1985). Freeze and Cherry (1979) also state that molecular diffusion is important only at low flow velocities. Figure 5-2 depicts movement by molecular diffusion. Note that no water movement is required for the dye to spread out (diffuse) in a direction tending to equalize concentrations in the blotter.

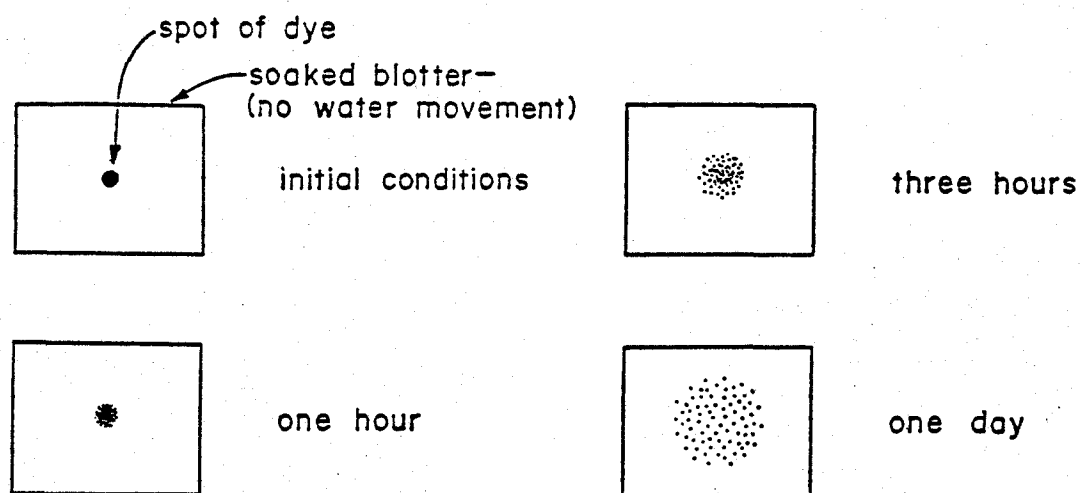


Figure 5-2. One dimensional example of movement by molecular diffusion (from Davis et al. 1985).

Hydrodynamic dispersion is the spreading which occurs both perpendicular to and in the direction of groundwater flow, of a water-soluble substance that is transported with the groundwater. The transit velocity of specific water-soluble substances may be higher or lower than the average groundwater velocity. Hydrodynamic dispersion is dependent on the velocity distribution in the medium and on molecular diffusion. Bear (1972) gives a quite detailed evaluation of hydrodynamic dispersion. Dispersion of a solute requires groundwater flow in a medium with a system of pores or channels. An example of hydrodynamic dispersion which is caused by unequal velocities of the ground water is shown in Figure 5-3. Tracer particles released at the same time and carried by the groundwater have different flow paths. This results in a more widespread distribution of tracer particles with time.

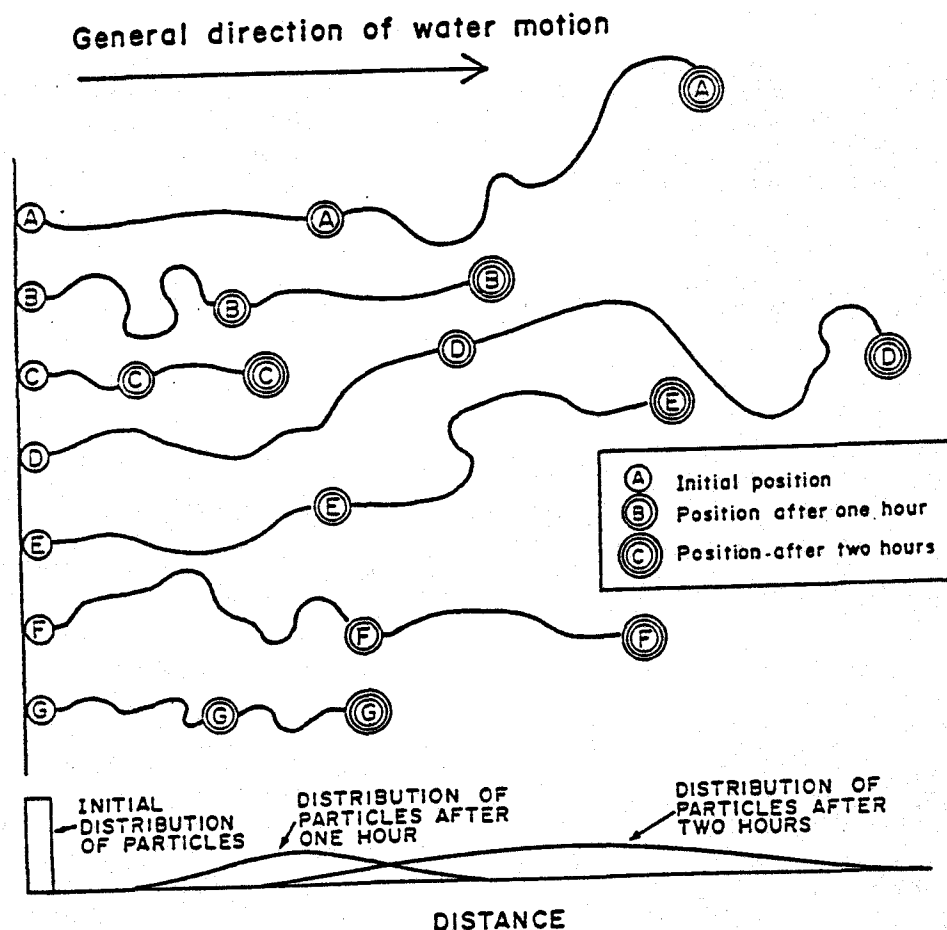


Figure 5-3. One dimensional example of hydrodynamic dispersion for tracer particles A - G (from Davis et al. 1985).

5.1.3 Tracer Methods

The initial step in conducting a tracer study is to collect as much hydrologic information as possible about the study site (Davis et al. 1985). This information should include homogeneity of the aquifer, layers present, fracture patterns, porosity, flow system boundaries, hydraulic gradient and hydraulic conductivity (Davis et al. 1985). This hydrologic information is used to assess the groundwater flow direction and velocity. Direction and velocity are usually estimated by the use of Darcy's law or by performing a preliminary tracer test (Davis et al. 1985). The second step is to determine the best tracer to use for the conditions and objectives at the site. The third step in conducting a tracer study is to determine the correct amount of tracer to be used. This amount is based on dilution expected, natural background concentration and detection limit possible for the tracer. The next step is to determine the correct tracer method.

One method involves the use of an environmental tracer which is a substance that exists in the soil before the investigation begins. This tracer can be artificial (man induced), semi-artificial (tritium) or natural (natural radioisotope). A good environmental tracer must be free from chemical reactions, such as ion exchange and precipitation and must not react with the medium (Fried 1975). Initial and boundary conditions must be known, chiefly by knowing the amount of tracer added and its history. The wetland system under investigation has the characteristic of a high organic matter content (large exchange capacity). Trace metals (environmental tracers) commonly react with the organic material, making them unacceptable in this situation.

Freeze and Cherry (1979) describe four main types of field dispersivity (tracer) tests. These are (1) single-well withdrawal-injection tests, (2) natural-gradient tracer tests, (3) two-well recirculating withdrawal-injection tests and (4) two-well pulse tests. In a natural gradient tracer study, the direction and velocity of the groundwater flow are very important (Davis et al. 1985). In the natural-gradient test, the tracer is introduced into the system and its migration is then monitored at one or more sampling points. Dispersivity values are obtained by fitting an analytical or numerical model to the experimental data. Davis et al. (1985) notes that:

"It is not at all uncommon to inject a tracer in a well and not be able to intercept that tracer in a well just a few meters away, particularly if the tracer flows under the natural hydraulic gradient which is not disturbed by pumping."

A tracer may be injected as a slug or as a continuous source input. Mixing of the tracer during injection is important in most types of tests and can be as simple as pouring it in the water to be studied (Davis et al. 1985). For shallow wells a plunger can be surged up and down in the hole or the tracer can be released through a pipe with many perforations. The ideal condition is to inject the tracer into the water instantaneously as a slug.

Taking field measurements of electrical conductance within, ahead of, and behind a tracer slug can minimize the number of field samples kept for laboratory analysis (Lee et al. 1980). Electrical conductance is the simplest and most inexpensive detection and analysis technique for ionic tracers and can be used as a break-through indicator (Slichter 1902). In the wetland system being studied, the naturally high ionic content may make electrical conductance data difficult to interpret.

5.1.4 Tracer Problems

Many water tracing attempts are unsuccessful. Aley and Fletcher (1978) have found that the major causes of failure are: (a) insufficient hydrological field work before the tracer is injected, (b) tracing attempts during low flow conditions, and (c) failure to allocate sufficient time for tracing effort. Davis et al. (1980) found that tracer test failures are most commonly a result of incorrect choice of tracers, insufficient concentrations of tracers and a lack of understanding of the hydrogeologic system being tested.

For a given head drop, expected travel time is a function of the distance squared, and therefore increases very rapidly with the distance. Davis et al. (1985) note that this relationship causes one of the most common errors in tracer tests, which is to conduct tests between points which are separated by too great a distance. Freeze and Cherry (1979) describe four main disadvantages to the determination of groundwater velocity by the direct tracer method.

- 1) Undesirably long periods of time, because of the fact that groundwater velocities are usually low, are normally required for tracers to move significant distances through flow systems.
- 2) Numerous observation points are usually required to adequately monitor passage of the tracer through the study area because geological materials are typically quite heterogeneous.
- 3) Because of (1), only a small and possibly nonrepresentative sample of the flow field is tested.

- 4) Because of (2), the flow field may be significantly distorted by the measuring devices.

The concentration of ion to be injected should be well above the natural background concentration level found at the test site and high enough to ensure detectable levels in observation wells (Davis et al. 1985). The ion concentration injected must also be kept low enough so that density effects do not effect flow of the tracer. An insufficient quantity of tracer will result in an unsuccessful trace: too much tracer wastes materials and can degrade water quality. Dilutions of a tracer in transit from injection to sampling wells are almost always at least tenfold for "slug" injections and dilutions of ten thousandfold are common (Davis et al. 1980).

Lenda and Zuber (1970) describe a method to estimate the adequate amount of tracer and carrier to be injected. The effects of adsorption and the uncertainty from whether the observation well is exactly at the center of the tracer path should be accounted for by the use of a safety factor on the order of 10. Skibitzke and Robinson (1963) used tracers to show that solid particles (sand grains) retard diffusion in a porous medium. Biggar and Nielson (1962) concluded that the mere presence of a tracer downstream from the point of injection is a poor indicator of the velocity of the fluid. Pore geometry, water content changes and the magnitude of the interaction between the tracer and the porous medium are important in determining an accurate estimate of the fluid velocity.

5.1.5 Bromide as a Tracer

There is no such thing as the perfect tracer but Davis et al. (1980) note that the ideal ground-water tracer is nontoxic, inexpensive, moves with the water, is easy to detect in trace amounts, does not alter the natural direction of the flow of water, is chemically stable for a desired length of time, is not present in large amounts in the water being studied and is neither filtered nor sorbed by the solid medium through which the water moves. Different types of tracers include: water temperature, solid particles, ionized substances (Br⁻), stable isotopes, radioactive tracers, organic dyes, gases, and fluorocarbons. Davis et al. (1980) report that some of the most useful general tracers are bromide, chloride, rhodamine WT, and various fluorocarbons. Most tracers have relatively limited or specialized uses.

Tracer selection should be based on purpose of the study, type of aquifer system, aquifer characteristics, natural background concentration of the ions in the groundwater, and analytical techniques available (Davis et al. 1985). In most

cases, anions are not affected by the aquifer medium (Davis et al 1985) but the characteristics of some aquifers will cause retention or exclusion of anions moving through the system.

Anionic tracers such as bromide (Br^-) and chloride (Cl^-) are particularly useful because of their low susceptibility to adsorption or ion exchange processes of natural aquifer materials. Bromide does not appear to be lost by precipitation, adsorption or absorption and is biologically stable (Schmotzer et al. 1973) and therefore can be considered a conservative tracer. Bromide offers one of the best possibilities as a general tracer for groundwater studies (Davis et al. 1980). Most bromide compounds also have relatively low toxicities. Davis et al. (1980) note that bromide samples, being nonvolatile can be stored indefinitely without concern for tracer loss to the atmosphere and sampling can be done using inexpensive air-lift pumps.

Davis et al. (1985) relate that bromide is perhaps the most commonly used ion tracer. These authors list advantageous characteristics of the use of bromide as a tracer as inexpensive, stable, low limit of detection, low background concentrations, low toxicity and no sorption. Bromide as a tracer is commonly injected as NaBr , CaBr_2 , or KBr .

The concentration of bromide in natural ground waters is roughly 1/300 that of chloride and usually <1 mg/L (Davis et al. 1980; Vinogradov 1959). Detection of bromide is relatively simple with a specific ion electrode which has a lower limit of detection of about 0.4 mg/L. If natural water has 30 mg/L of chloride (suggesting the natural presence of 0.1 mg/L of Br^-) and if the bromide tracer is introduced with a concentration of 1000 mg/L, then a dilution factor of 10^4 is possible before it is masked by the natural background (Davis et al. 1980). Concentrations in our introduced chloride tracer should not exceed about 3000 mg/L because of increased density of the solution (Davis et al 1980). Since the halides bromide and chloride behave similarly (Bohn et al. 1985) it is thought that bromide should not exceed the 3000 mg/L concentration.

Davis et al. (1985) state that an advantage of anions, such as bromide, used as tracers is that they do not decompose and are not lost from the system. This statement is generally true but under certain circumstances anions such as bromide may be affected by anion exclusion and/or anion exchange. As anions move through soil they do not come in contact with all of the soil water. This is termed anion exclusion and occurs in response to the fluid flow rate and the fact that water near negatively charged soil surfaces is relatively immobile. The result of this exclusion is that anions can move through the soil faster than one would predict on the basis of uniform association with all the soil water (Smith and Davis 1974). The association between anion exclusion and cation exchange capacity is strong.

Anion exclusion is a manifestation of the unequal ion distribution in the diffuse double layer surrounding charged colloid surfaces (Bohn et al. 1985). Factors affecting anion repulsion (exclusion) include: 1) anion charge and concentration, 2) species of exchangeable cation, 3) pH, 4) presence of other anions, and 5) nature and charge of the colloid surface (Bohn et al. 1985). Thomas and Swaboda (1970) suggest that in soils with high cation exchange capacities anion exclusion causes anions to move much faster than they would if no interaction with clays were to occur. This suggestion agrees with the theory that anion exclusion is a function of negative charge.

Grim (1968) mentions two types of anion exchange in clay minerals. One is the replacement of hydroxide ions and the other factor is related to geometry of the anion. The geometry of the bromide anion does not fit that of silica tetrahedral sheets, thus it cannot be so absorbed. Anion exchange would take place around the edges of the clay minerals, not on the basal plane surface. A factor which complicates anion exchange studies is that any free or exchangeable iron, aluminum or alkaline earth elements present in the clay may form insoluble salts with the anions (Grim 1968). Mattson (1929) has shown that the adsorption of anions was found to be negative in neutral and alkaline solutions and that as pH decreases the capacity of clay minerals for holding anions increases. The interaction of cations with clays is much more frequent than with anions. Berg and Thomas (1959) found that sulfate and chloride anions are adsorbed in soils high in kaolin clays and aluminum and iron oxides. Chloride will desorb readily at pH values found in most field conditions, but at low pH values, chloride ions were not easily desorbed. They also found that sulfate is held much more tightly to these soil types than is chloride.

The study site is in the headwaters area of the Blackfoot River and bromide toxicity was a concern. Alexander et al. (1981) studied the effect of sodium bromide on Fathead minnows. Sodium bromide has a low toxicity to Fathead minnows with the average LC₅₀ (lethal concentration for 50% of the test population) for a 96 hour period being 16,479 mg/L. Tests by Barnes et al. (1981) show that the mean number of organisms (thirty-five species of algae and zooplankton were studied) increased with time due to the addition of 1000 ppb potassium bromide and nutrients to the system. Species diversity decreased slightly in this situation. Schmotzer et al. (1973) reported that bromide has a very low toxicity in humans at 50-100 mg of bromide/100 ml of blood. This translates to a human having to drink 12 liters of 200 mg/L bromide to be toxic. The oral LD₅₀ (lethal dose) in rats is 3.5 g/kg (Merck Index 1983).

Martin et al. (1956) concluded that plant tolerance to bromide is quite variable. Carrot tops contained 2.5% (25,000 ppm) bromide with no reduction in growth but citrus trees containing 0.2% (2000 ppm) demonstrated reduced plant growth.

The desirable characteristics of using bromide as a soil water tracer were demonstrated by Onken et al. (1977) to be easy detection, unlikely contamination of the environment and lack of reaction with soil and soil constituents. Schmotzer et al. (1973) conducted a fairly extensive study of using bromide as a groundwater tracer and found bromide to successfully fulfill the requirements devised by Schmotzer to be as close as possible to being the ideal tracer. The favorable characteristics of bromide include low toxicity, high sensitivity of detection, little loss through precipitation, adsorption and absorption, high stability, low background concentrations, small sample size requirements (post sampling activation analysis), low cost, government approval is relatively easily attained and bromide is biologically stable.

Vinogradov (1959) found that bromide correlated well with iodide content and both bromide and iodide content was proportional to the amount of organic material in the soils studied. The concentration of bromide is greater in humic soils and there is practically no dependence of chloride content on soil organic matter content. The studies by Vinogradov (1959) suggest that bromide is sorbed by peats and that the amount sorbed decreases somewhat with aging of the peats. Vinogradov (1959) found that as soil organic carbon content increased the bromide and iodide content increased, but no effect was observed on the chloride content.

Smith and Davis (1974) found that bromide is a good tracer for mimicking the movement of nitrate (NO_3) through subsoils. Merrill et al. (1985) used bromide as KBr to trace $\text{NO}_3\text{-N}$ movement and indicate water flux in a study to develop an understanding of plant growth response to soil thickness over sodic minespoils. Onken et al. (1977) used sodium bromide and sodium nitrate to show that nitrate and bromide move together in the soil profile. They note that both nitrate and bromide are readily absorbed by plants but the rates of removal from soil are different.

Tennyson and Settergren (1980) used a sodium bromide tracer to evaluate percolating water and ion movement in an irrigation saturated surface soil. Background levels of bromide in soil water, groundwater, and precipitation were measured and bromide movement was quantified by soil water sampling and post-sampling neutron activation analysis. They suggested that laboratory measured hydraulic conductivity was not adequate in evaluation of the site because the bromide tracer moved through the soil much more rapidly than the hydraulic conductivity suggested. Peak concentrations of the bromide tracer moved through 0.9 m of these

soils within 3.75 hours after field application. Tennyson and Settergren (1980) found indication of bromide retention occurring because bromide concentrations above background levels were present in the soils studied three weeks after application of the tracer.

5.1.6 Wetland Hydrology

Wetland systems can be considered quite heterogeneous. Figure 5-4 is shown to exemplify this point. Notice that not only is there seven orders of magnitude of hydraulic conductivity variation reported in the literature shown but there is often quite a large range of conductivity within a study site. Freeze and Cherry (1979) and Davis et al. (1985) note that a problem

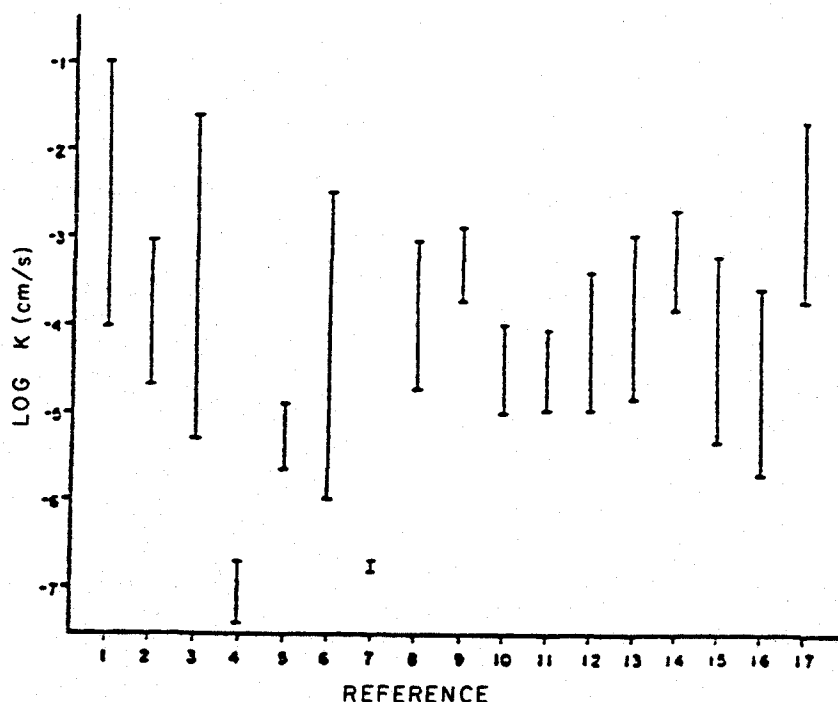


Figure 5-4. Ranges of published field data on K (hydraulic conductivity) of peat: 1. Baden and Eggelsman (1961, 1963, 1964); 2. Eggelsman and Makela (1964); 3. Boelter (1965); 4. Ingram (1967); 5. Galvin and Hanranahan (1968); 6. Romanov (1968); 7. Sturges (1968); 8. Dowling (1969); 9. Irwin (1970); 10. Yamamoto (1970); 11. Knight et al. (1971); 14. Paivanen (1973); 15. Galvin (1976); 16. Dasberg and Neuman (1977); 17. Chason and Siegel (1986). Adapted from Chason and Siegel (1986).

arises in determining the direction of water movement and travel time when local differences in hydraulic conductivity amount to several orders of magnitude. The following paragraphs describe some of the findings of specific authors in regard to wetland characteristics.

Hydrogeologic factors (Winter and Carr 1980) that affect wetland ground water flow include:

- 1) geometry of the geologic framework through which ground water flows, including flow boundaries;
- 2) hydraulic conductivity of the geologic materials, including anisotropy -- the ratio of vertical to horizontal hydraulic conductivity;
- 3) recharge and discharge of the ground water system.

In particular, Winter and Carr (1980) found three major interrelationships between wetlands and ground water:

- 1) some wetlands appear to recharge ground water;
- 2) some wetlands are flow-through types where ground water enters one side and surface water seeps into the ground on the other side;
- 3) some wetlands are discharge areas for ground water.

These interrelationships are further complicated by changing throughout the year.

One of the more important physical properties that affect the hydrologic features of bog areas is the hydrologic conductivity of the peat soil horizons (Boelter 1965). Dai and Sparling (1973) suggested using the piezometer method to determine the velocity of flow from the hydraulic conductivity. Ingram et al. (1974) used field experiments to find that estimates of hydraulic conductivity in humified peat increased with head, which is not in accordance with Darcy's law. It is suggested that Darcy's law may only apply in peat of low humidification. Boelter (1965) used field measurements of hydraulic conductivity (K) and found they covered a wide range of values. Chason and Siegel (1986) found that horizontal K was significantly greater than vertical K.

Sturger (1968) studied the hydrologic properties of a mountain bog in Wyoming. Undecomposed surface peat had the lowest bulk density, 0.160 g/cc, and much of its volume consisted

of large voids which emptied at suctions of less than 0.10 bar. Greater suctions were required to drain the smaller pores of the higher bulk density, decomposed material found at depth.

In 1969 Boelter found that the classification of peat materials based on degree of decomposition as measured by fiber content (> 0.1 mm) and bulk density can relate significant information about the hydraulic conductivity, water retention, and water yield coefficient. Regression equations were employed to determine a range in hydraulic conductivity and other physical properties for fibric, hemic and sapric peat materials (Table 5-1). In studying a Minnesota spring fen-raised bog complex Chason and Siegel (1986) found that hydraulic conductivity, bulk density and humicity all vary widely through the peat profile and in most cases were not mutually dependent. There are two major hydrologic zones within a peat soil column (Ingram 1983):

- 1) the acrotelm, a thin upper aerated zone of fluctuating water conditions composed of undecomposed dead and live vegetation,
- 2) The catotelm, an underlying anaerobic zone in a constant waterlogged state composed of more humified peat.

Some authors (Dasberg and Neuman 1977; Ingram 1967; Sturges 1968) note erosion channels or fractures at depths within wetland systems. The results of field and laboratory studies show that the properties of peat change drastically when it becomes partially desaturated (Dasberg and Neuman, 1977). Two layers result from this change, a permanently saturated layer below the zone of water table fluctuation, and an overlying partially unsaturated layer with properties that vary with depth. The saturated peat layer studied had a high porosity (90%); organic matter content (60% by weight); specific yield (20 to 30%); and a very high specific storage (0.7×10^{-3} , to 1.7×10^{-3}). The unsaturated peat layer was found to have a high bulk density, up to 0.8 g/cc near the surface. The hydraulic conductivity of the unsaturated layer is generally higher than 1.0 mm/h (3.0×10^{-5} cm/s), but much more variable than the saturated peat layer. In summer, shallow cracks often develop which may increase the hydraulic conductivity by several orders of magnitude in the unsaturated surface layer.

Ingram (1967) described a "water track" as a mire surface feature having a higher rate of water movement due to more steeply inclined water tables and/or peat with higher hydraulic conductivity. These water tracks are associated with the presence of more eutrophic plant communities and are sites of greater ion supply. Erosion channels occur both on the surface and at depth in peat deposits with the associated vegetation often being unlike the surrounding vegetation. Subterranean erosion tunnels are frequently encountered in blanket bogs but

Table 5-1. Range of important physical characteristics of fibric, hemic, and sapric peat materials from northern Minnesota bogs (From Boelter 1969).

Organic Material	Bulk density (g/cc)	Total porosity (%)	0.1 bar H ₂ O content (%)	Hydraulic conductivity (10 ⁻⁵ cm/sec)
Fibric	<.075	>90	<48	>180
Hemic	.075-.195	85-90	48-70	2.1-180
Sapric	>.195	<85	>70	<2.1

little is known of these concealed free drainage systems (Ingram 1967). Wetland hydraulic conductivity measured in situ by the piezometer method (Sturger 1968) at depths of 46 and 91 cm was 23.9×10^{-3} and 16.1×10^{-3} cm per day respectively. Two of five piezometers at the 91 cm depth showed much higher hydraulic conductivity, indicating the piezometers terminated in or near a fissure. The fissures running through the deeper peat are filled with water under positive pressure and there are no surface physical features to indicate fissure location. Overall wetland hydrologic characteristics are found to be complicated by the fact that they vary greatly between wetlands, within wetlands and with time.

Tracers have seldom been applied to wetland hydrology, "primarily because of the difficulty in choosing an appropriate tracer and the disturbance of the experimental area which results from frequent sampling (Girts 1986)." Knight et al. (1971) successfully used a tritium tracer to reveal variable patterns of movement in a wetland under different drainage conditions. The flow rate variation was found to be attributed to factors within the peat rather than to analytical procedure. Girts (1986) successfully used bromide and chloride tracers to yield flow rates within the range of published values (see Figure 5-4).

Bowmer (1987) used bromide and dye tracers to study a man made wetland used to treat sewage. In this study theoretical detention time of effluent was predicted using the pore volume of the system. Preferential flow paths and "dead zones" were found using the tracers and a substantial proportion of the effluent was found to travel through the system faster than predicted by the theoretical retention time. The hydrology of the system was further complicated by occasionally high evapotranspiration rates and diurnal changes in water consumption.

5.2 MATERIALS AND METHODS

5.2.1 Site Instrumentation

Thirty-eight shallow small diameter wells (piezometers) were installed throughout the study area during the spring and summer of 1987. Twenty-eight of these wells are paired, with one well shallow and one deep resulting in 24 total well sites (Figure 4-1). The shallow wells were completed to a total depth between 1.2 to 1.5 m (4 to 5 ft) and deep wells to a total depth between 1.8 and 4.9 m (6 and 16 ft). Wells were constructed of 2.5 cm (1 in) diameter PVC pipe and Timco 0.025 cm (0.01 in) slotted, capped PVC screen. All wells were completed with a 20-30 grain sand pack and bentonite pellets. Figure 5-5 depicts a typical paired well site construction. All of these wells were developed by surging and overpumping.

Well holes were constructed using 5.1 cm (2 in) diameter Giddings soil core barrels hand driven into the ground. A 7.6 cm (3 in) silage auger (Arts 1987) modified with a core cutter/holder was used to obtain soil samples in the 0 to 1.2 m (0 to 4 ft) depth at each well site. An additional 32 auger holes were excavated near the highway culvert outflow to a depth ranging between .5 and .9 m (1.6 and 3 ft). These additional auger holes were positioned (Figure 5-6) so as to facilitate tracer sampling. A Parshall flume equipped with a Stevens Type F recorder was installed June 15, 1987 on Swamp Gulch below the acid mine seep to quantify flow rates into the wetland. A Belfort weighing precipitation gauge was also installed at the site in June 1987. The position of both the flume and precipitation gauge are noted in Figure 4-1.

5.2.2 Aquifer Characteristics

Porosity, bulk density and particle density were determined on soil samples obtained during excavation of auger holes. Bulk density was determined using the Core Method described by Blake and Hartge (1986a). The Pycnometer Method was used to determine particle density (Blake and Hartge 1986b). Total porosity, n , was calculated with the following equation (Danielson and Sutherland 1986):

$$n = (1 - pb/pp), \quad (\text{Eq. 5-2})$$

where pb = bulk density
 pp = particle density

5.2.3 Hydraulic Conductivity

Saturated hydraulic conductivity was measured in the field using both auger hole and seepage tube (piezometer) methods. A

variable head method modified from the hydrostatic time-lag method of Hvorslev (1951), as reviewed by Cedergren (1977), was used to calculate saturated field hydraulic conductivity. This method is based on the rate at which water rises or falls in a hole after a known volume (slug) has been removed or added. In the case of the auger holes a bailer was used to remove a volume of water and the rate of water rise was measured using equipment designed similar to that described by Beers (1983). For the piezometers a volume of a solid steel rod was added and the rate of water fall was measured using an electric tape.

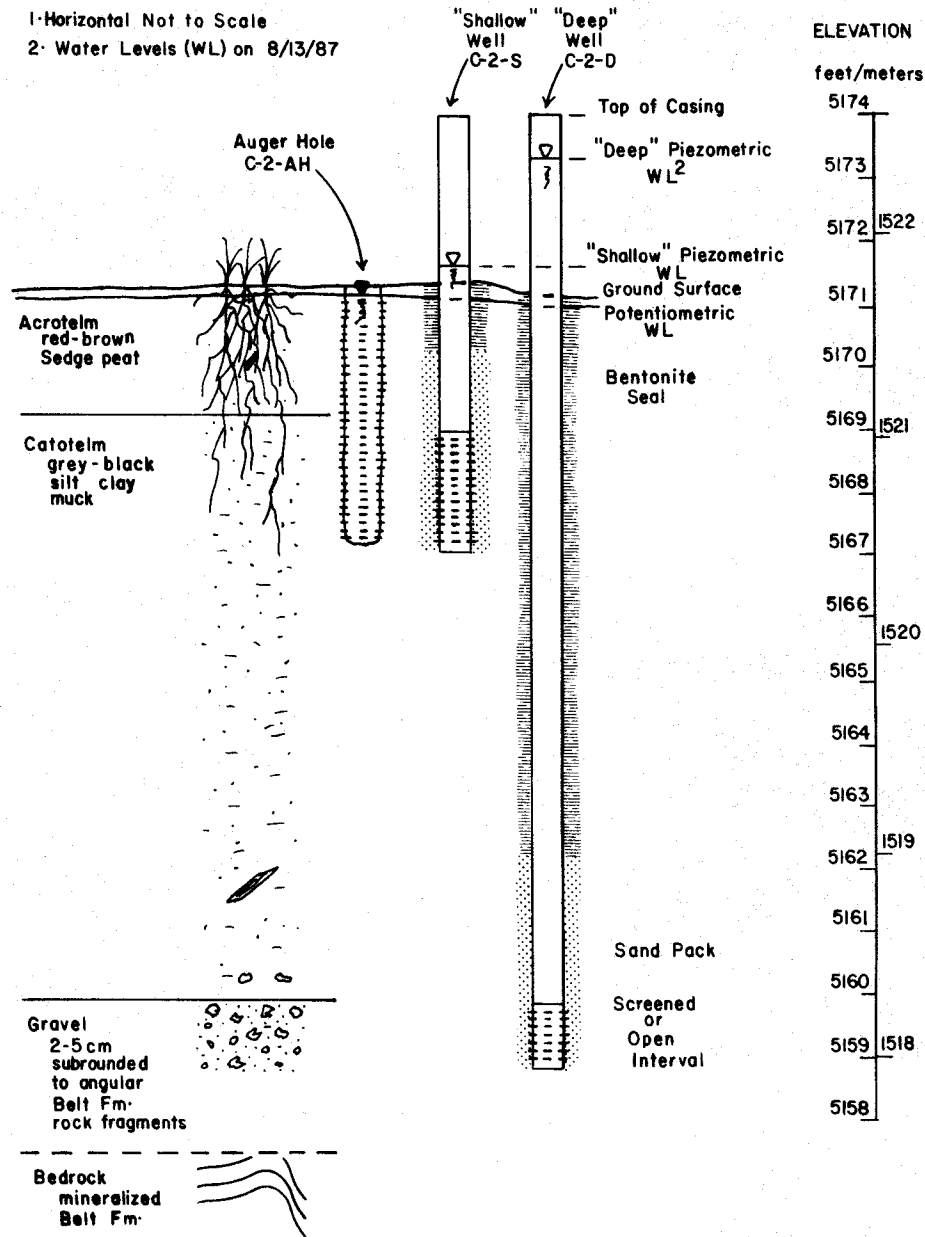


Figure 5-5. Example of study site instrumentation and lithology.

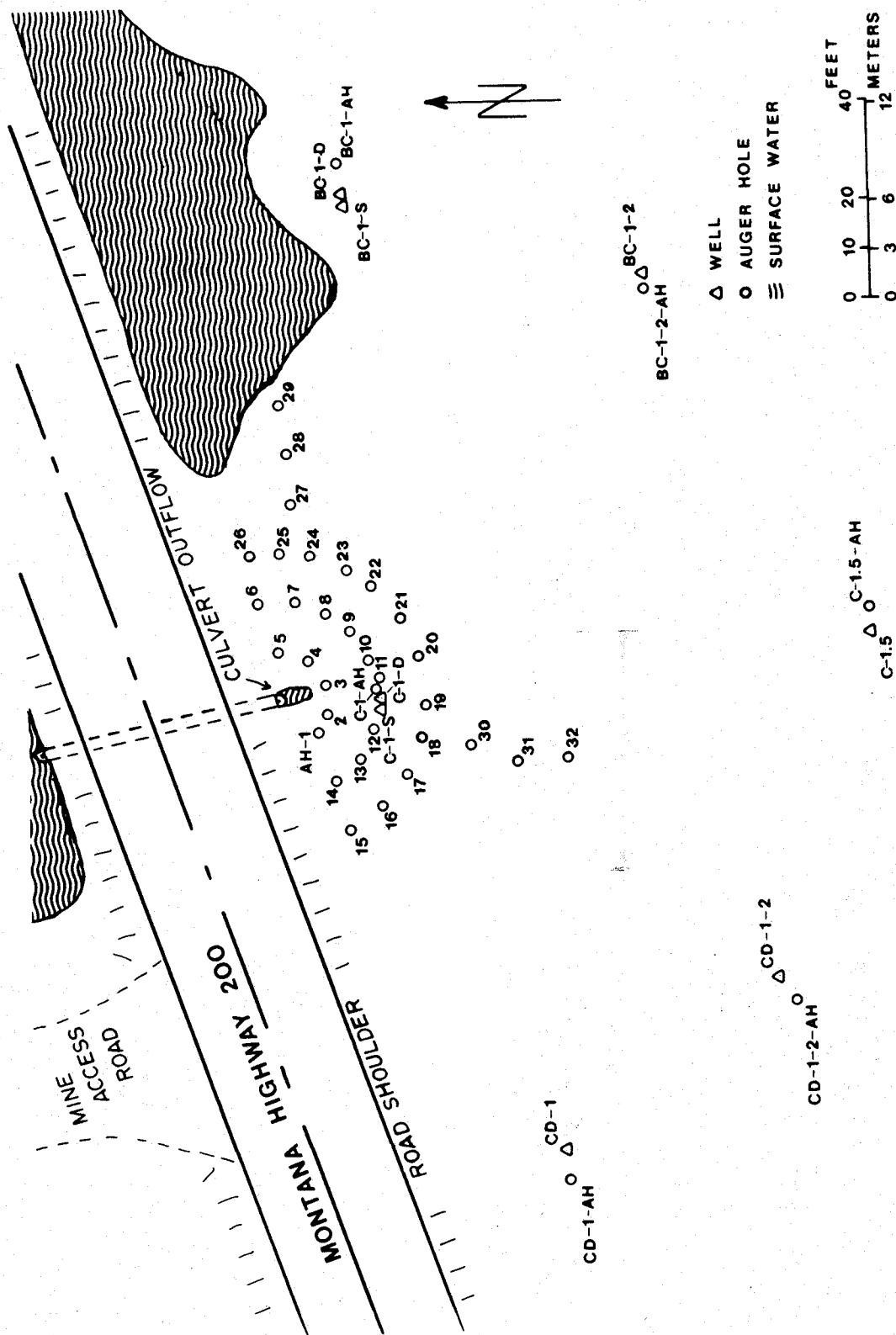


Figure 5-6. Map of auger hole grid.

Hydraulic conductivity was calculated for auger holes and piezometers, respectively, using the following equations:

$$K = \frac{R}{16DS} \times \frac{(h_2 - h_1)}{(t_2 - t_1)} \quad (\text{Eq. 5-3})$$

and

$$K = \frac{r^2}{2L} \ln \left(\frac{L}{R} \right) \frac{\ln(h_1/h_2)}{t_2 - t_1} \quad (\text{Eq. 5-4})$$

Where: R = radius of cavity
 D = depth of cavity below static water table
 S = shape factor coefficient
 h₁ = head at time one
 h₂ = head at time two
 t₁ = time one
 t₂ = time two
 r = radius of the well
 L = length of screen

Hydraulic conductivity was converted to velocity (V) to obtain a preliminary estimate on which to base the tracer test sample timing. These velocities determined from hydraulic conductivity also facilitate comparison with velocities determined by the tracer test. The formula used to make the K-V conversion is shown in equation 5-1.

5.2.4 Tracer Experiment

A preliminary dye tracer test was run using fluorescein dye. This experiment showed preferential flow direction, helped determine the tracer input point and resulted in an even better idea of flow rates near the tracer input point. In this experiment approximately 7 grams of fluorescein powder was added to 18.9 liters (5 gal) of water, mixed and introduced at different points along the mine drainage flow path. Tracer movement was monitored visually.

Bromide was chosen as the tracer in the experiment for use in determining water flow velocities. Prior to adding bromide to the wetland, background (natural) bromide concentrations were measured. These background concentrations were measured using the bromide specific ion electrode as described below and by the EPA (1979) titrimetric Method 320.1. In the titration method the EPA (1979) found bromide recoveries from 83% to 99% and standard deviations from ± 0.13 to ± 0.44 for sample concentrations ranging from 0.3 to 20.3 mg/l. The titration method as used in this study showed 100% accuracy based on two laboratory spikes.

The natural gradient tracer experiment utilized 208 liters (55 gal) of wetland water at a concentration of 1000 mg/L bromide. This water was added at the point of highway culvert outflow at 10:30 AM on September 9, 1987 (time zero). Fluor-

escein dye (18.9 liters - 5 gal) was added at this time as a visual aid in bromide sampling. The tracer was added to the system as an extended slug input at a rate of 0.39 l/s (0.625 gpm).

Flow of the tracer plume was monitored by sampling the auger holes and wells and analyzing for bromide. Water sampling was done using portable peristaltic pumps. Care was taken in sampling so that a representative water sample was acquired, but not so much water that the flow field was greatly affected. This care usually amounted to pulling one well volume to rinse the sample bottle three times and taking the 100 ml sample with the second well volume.

Water samples were then analyzed for bromide in the field using a bromide selective ion electrode in conjunction with a pH/millivolt (mv) meter. Calibration curves (mv vs mg/L) were generated using standards measured under the same conditions as the samples (Orion Research 1982). Using the bromide electrode the concentration range is from 0.4 to 79,900 mg/L with a $\pm 2\%$ reproducibility.

The sampling intervals at the start of the experiment were nearly continuous near the input point. Sampling had decreased to once every other day, by September 14 (124 hours). From September 26 (412 hours), to the end of the field experiment, November 21, 1987 (1755 hours), sample sets were taken once each week. Water sampling was terminated after the tracer plume had passed a particular well or auger hole. November water sampling was hampered by freezing conditions. The passing of a tracer plume was determined from the bromide breakthrough curves.

Bromide breakthrough curves are plotted as bromide concentration in milligrams per liter (mg/L) vs time in hours. Concentrations were generated from the sample analysis data from a particular site and calibration curves from the time of sample analysis. Times represent the time a particular sample was taken from the wetland. The average velocity of water flow was taken to be the distance from the tracer input point to a particular well or auger hole divided by the time of peak concentration for that site breakthrough curve.

5.3 RESULTS AND DISCUSSION

5.3.1 Aquifer Characteristics

Field evaluation of auger hole and well cores showed a fairly distinct boundary between the upper, undecomposed (fibric) organic matter (sedge peat) and the underlying moderately decomposed (hemic) organic matter. Little distinction was made between moderately and well decomposed zones which are grey to

black organic matter muck with occasional sand and/or gravel (<5 cm) stringers and rotted wood pieces (<10 cm). The horizonization scheme used distinguishes between two soil types 1) upper, undecomposed organic matter (acrotelm), and 2) underlying muck (catotelm) (Figure 5-5). The upper undecomposed zone is aerated and subject to fluctuating water conditions and water is transmitted comparatively faster in the upper zone compared to the underlying muck zone which is anaerobic and constantly waterlogged. In the wetland being studied, the acrotelm thickness ranges from 0.24 m (0.8 ft) to 1.04 m (3.4 ft) with an average of 0.58 m (1.9 ft) (Figure 5-7) and catotelm thickness ranges from 0.58 m (1.9 ft) to 4.30 m (14.1 ft) with an average of 2.16 m (7.1 ft) (Figure 5-8). This information is summarized in Table 5-2. Note from figures 5-7 and 5-8 that the greatest thickness for both zones are near the highway culvert outflow. These greater thicknesses may be due to sediment deposition from Swamp Gulch or metal precipitation originating from AMD water.

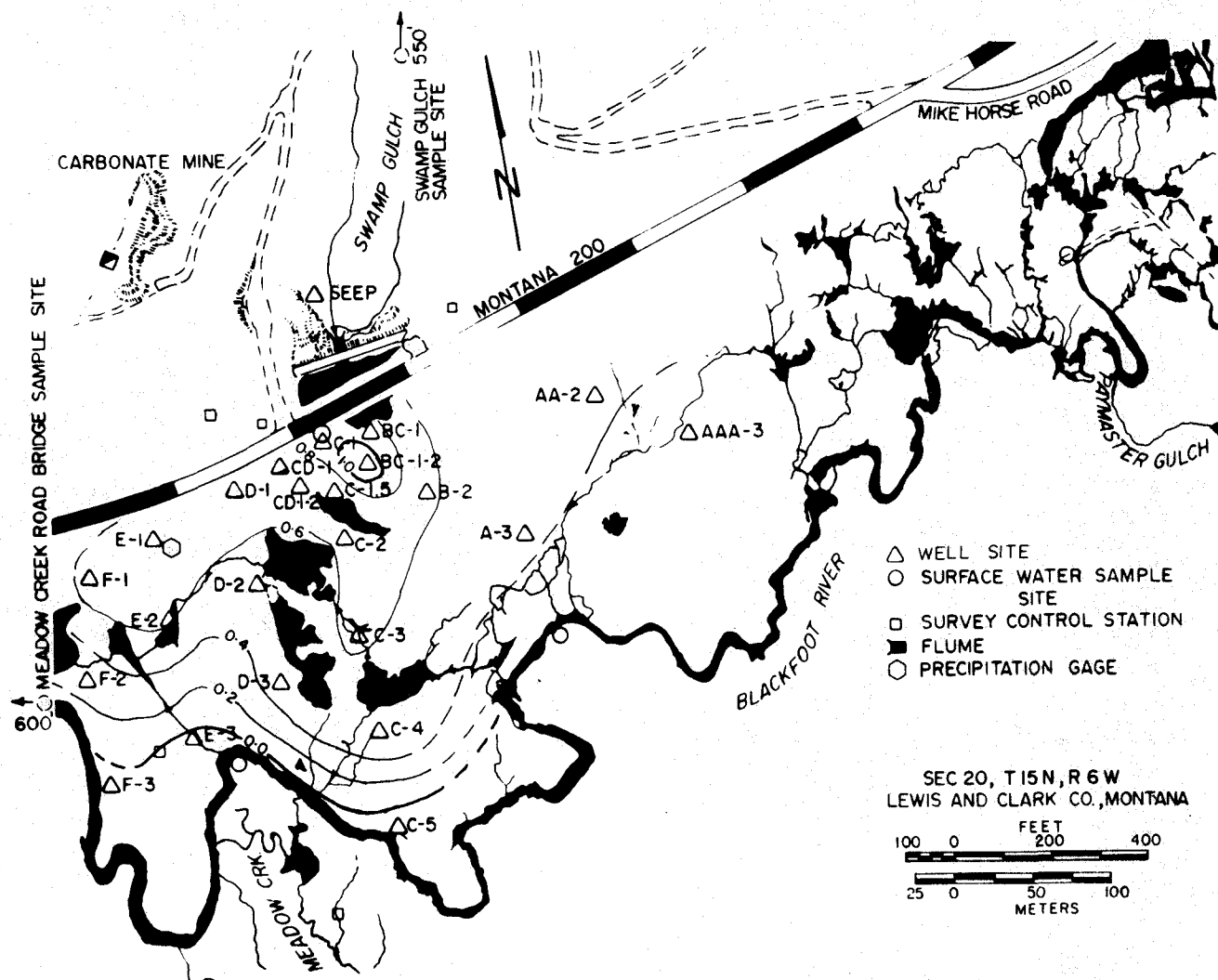


Figure 5-7. Acrotelm isopach map with 0.2 m (0.7 ft) contours.

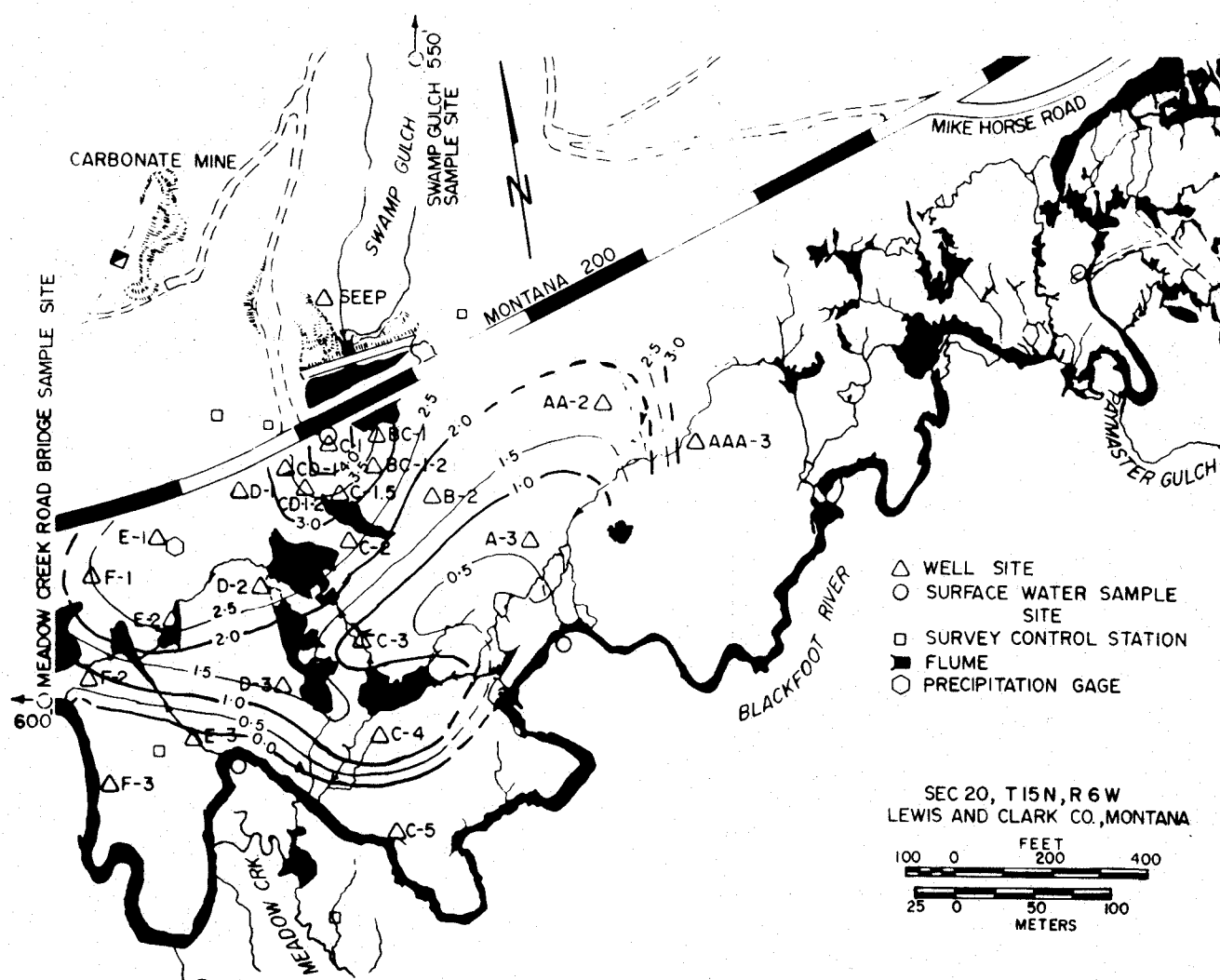


Figure 5-8. Catotelm isopach map with 0.5 m (1.6 ft) contours.

Both sediment deposition and metal precipitation are indicated in the wetland soils data (Appendix C). Sediment deposition in the acrotelm near the highway culvert outflow site, C-1 (0 to 91 cm), is indicated by a higher sand content (37%) and lower organic carbon content (12.4%) than the other three wetland acrotelm sites analyzed for these parameters. Acrotelm metal loading at this site is indicated by the highest observed iron content in the study site at 415,000 ppm (41%). The sediment deposition and metal loading at site C-1 has had an interesting effect on cation exchange capacity (CEC) and anion exchange capacity (AEC). The low organic carbon content and high sand content of the acrotelm at site C-1 have likely contributed to the observed low CEC of 4.9 meq/100g. The resulting increase in CEC with depth at this site is the opposite of that expected and

Table 5-2. Wetland stratigraphy and well completion data.

Site	Well ID-BGS*	Associated Auger Hole ID-BGS	Screen Length	Undecomposed OM+ Thickness or ID-BGS	Decomposed OM Thickness	Depth to Gravel BGS / Core II)
Meters						
AAA-3-D	3.45		0.30	0.24	3.08	3.32 4.57
AAA-3-S	1.51		0.58			
AA-2-D	3.64		0.58	0.46	1.83	2.29 2.56
AA-2-S	1.32		0.58			
A-3	1.53		0.52	0.46	0.58	1.04 1.95
B-2-D	2.62	0.92	0.58	0.61	1.83	2.44 3.66
B-2-S	1.27		0.58			
BC-1-D	2.44		0.58	0.85		
BC-1-S	1.22		0.58			
BC-1-2	1.25	0.81	0.58	1.04		
C-1-D	4.89	1.01	0.58	0.91	4.29	5.21 5.25
C-1-S	1.22		0.58			
C-1.5	1.25	0.87	0.58	0.61		
C-2-D	3.78	1.22	0.30	0.61	2.76	3.43 3.78
C-2-S	1.31		0.58			
C-3-D	2.12		0.58	0.61	0.91	1.52 2.59
C-3-S	1.24		0.58			
C-4-D	1.94		0.46	0.46	1.39	1.84 3.11
C-4-S	1.22		0.58			
C-5-D	3.11		0.58	0.00	0.00	1.99 3.11
C-5-S	2.00		0.58			
CD-1-2	1.24	0.94	0.58	0.58		
CD-1	1.23	1.16	0.58	0.61	2.68	3.29 4.27
D-1-D	3.99	1.04	0.58	0.61		
D-1-S	1.31		0.58			
D-2-D	2.44		0.58	0.46	2.53	2.99 3.66
D-2-S	1.22		0.58			
D-3	1.28		0.58	0.46	1.55	2.01 2.44
E-1	1.26	0.91	0.58	0.61	2.67	3.28 3.66
E-2	1.37	0.80	0.58	0.61	2.74	3.35 3.66
E-3	1.27		0.58	0.00	0.00	1.07 1.98
F-1-D	2.52	0.91	0.58	0.61	2.50	3.11
F-1-S	1.75		0.58			
F-2-D	2.13		0.58	0.30	0.91	1.22 2.23
F-2-S	1.10		0.58			
F-3	1.26		0.58	0.00	0.00	0.12 1.22
SEEP-D	2.20		0.58	0.00	0.00	
SEEP-S	0.50		0.58			

Notes: * Total Depth - Below Ground Surface
+ Organic Matter

observed at the other wetland sites measured for CEC. The AEC exhibits an inverse relationship with CEC and decreases with depth at site C-1. The high AEC (90 meq/100g) at site C-1 may have the effect of retarding the bromide anion flow velocity by sorption of bromide on the exchange sites.

Values for bulk density, particle density and porosity were measured on three surface soil samples (acrotelm, 0 to 30 cm depth). Undisturbed samples were not possible below the 30 cm depth due to the sampling methods used and the nature of the wetland soils. The average bulk density (0.26 g/cc) is higher than that found by other studies at this depth in a wetland (Sturges, 1968 - 0.184 g/cc; Chason and Siegel, 1986 - 0.06 to 0.165 g/cc and; Boelter, 1969 - <.075 g/cc). The distinctness between the measured bulk density and that from other studies may be due to differing vegetation types and/or soil compaction during the sampling procedure.

Particle density values range between 2.077 and 2.942 g/cc with an average of 2.595 g/cc. This average density is less than the 2.65 g/cc used as an approximation for many mineral soils (Danielson and Sutherland 1986), but higher than the usual case for humus which is <1.5 g/cc (Blake and Hartge 1986b). The relatively intermediate, average particle density is presumed to be the effect of the high organic matter content and high heavy metal loading in the wetland.

Bulk density and particle density were used to find the total porosity (equation 5-2). Total porosity ranged from 87.35% to 94.27% with an average of 90.30%. This average porosity is similar to that found by Boelter (1969) which was >90% (Table 5-1), but higher than that calculated by Girts (1986), 66.1% to 76.1%. The values found by Girts (1986) were for partially and greatly decomposed peat and assumed no water was held within plant tissues. Boelter (1969) noted that total porosity decreased with degree of decomposition or depth in a wetland.

Effective porosity is less than total porosity because in small pores the retention forces are greater than the weight of water. Effective porosity in practice may be considered equal to the specific yield of an unconfined aquifer (Kruseman and Ridder 1983). Generally, specific yield is the water yielded by gravity drainage from water bearing material (Lohman 1979). Boelter (1965) found specific yields in undecomposed peat ranged between 0.79 and 0.33 cc/cc and that moderately to well decomposed peat ranged from 0.22 to 0.10 cc/cc. The averages of the above ranges, 0.54 and 0.186 cc/cc respectively, were used as the effective porosity for converting field hydraulic conductivities to velocities. The decrease in effective porosity with decomposition is appropriate because the more decomposed peat materials have more small pores which are not easily drained. The screens of the deeper wells were completed within or near the

gravel zone lying below the wetland. The effective porosity of deep wells completed in gravel, as used in conductivity to velocity conversions, was 0.25 cc/cc (Driscoll 1986).

Results of water level measurements in both wells and auger holes (Appendix B-4) indicate both upward and downward vertical hydraulic gradients in parts of the wetland. The artesian gradient (upward vertical) can be explained by two different arguments: 1) a confined aquifer system at depth, or 2) the wetland serves as a groundwater discharge area. Hydraulic conductivities which will be discussed later, suggest that the wetland system is capable of transmitting water throughout the soil profile (not confined). Therefore, the wetland is considered both a groundwater discharge and recharge area. Field observations, such as at the culvert outflow, suggested water flowing into the wetland also seeped into the soil profile and became groundwater. Surface flow channels which would disappear into the ground and apparently to reappeared down gradient were also noticed in the wetland. Situations where wetlands function as groundwater discharge, recharge and flow-through areas are not uncommon (Winter and Carr 1980).

Deep and shallow piezometric maps and a potentiometric map are shown in Figures 5-9, 5-10, and 5-11, respectively. A general comparison of these maps reveals that the hydraulic gradient does not change between water surfaces measured, and there is a hydrologic and topographic (Figure 4-1) high near the culvert outlet. The average hydraulic gradient across the wetland where the hydraulic conductivity tests were performed is 0.025 m/m. The change in head within a shallow and deep well and auger hole nest is generally less than 0.30 m (1 ft). This relationship can be seen in Figure 5-5. Overall piezometric surface levels stayed constant throughout the period measured. The potentiometric surface fluctuated approximately 0.30 m for short periods of time (days) during the tracer test. These fluctuations may be attributed to increased water use by phreatophytic plants (Mayboom 1967) during period of high air temperatures and the lack of evapotranspiration during the winter. This is further evidenced by slight diurnal fluctuations in water flow rate at the Swamp Gulch flume. The rate of water flow into the wetland from the culvert exhibited little change during the tracer study as indicated by the flume recorder charts (Appendix B-1). Precipitation during the tracer study was very low (Appendix B-2).

5.3.2 Hydraulic Conductivity

Hydraulic conductivity tests were run on auger holes and wells in the area of expected metal loading and expected tracer movement. This area was determined from water chemistry data and hydraulic gradients in the wetland. Field determined hydraulic conductivity values and the calculated water velocities for each

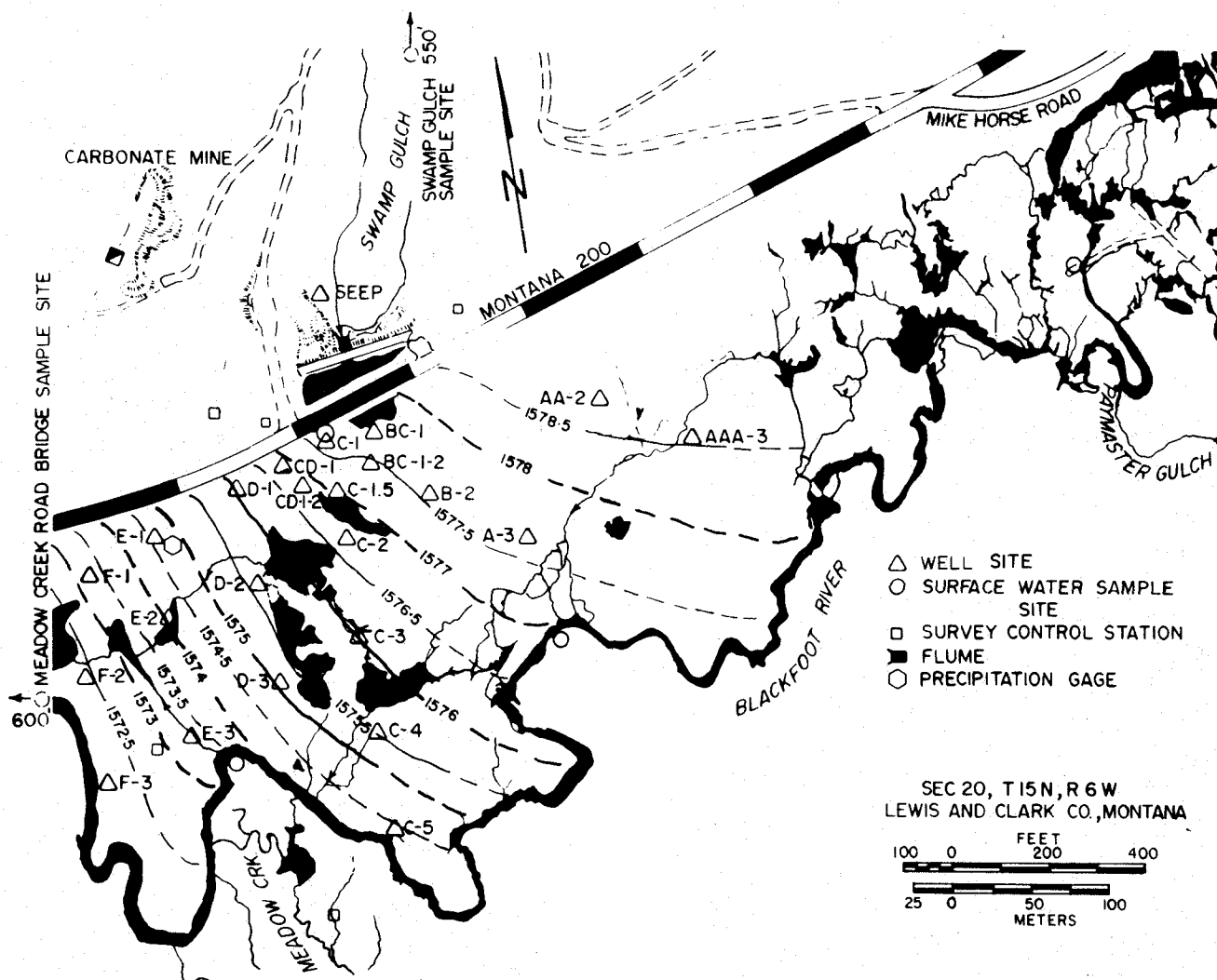


Figure 5-9. Deep piezometric map on 7/27/87 with 0.5 m contours.

auger hole and well tested are given in Table 5-3.

When field hydraulic conductivity values were grouped, some differences were found between auger holes and wells (Table 5-4). Lowest conductivity values were found in the shallow wells with their screened intervals within the catotelm (well decomposed organic matter, muck). Highest conductivity values were found in the deep wells with their screened intervals within both the catotelm and the underlying gravel. Auger holes which have their open intervals within the acrotelm (undecomposed organic matter) had conductivities intermediate in comparison to the wells (Table

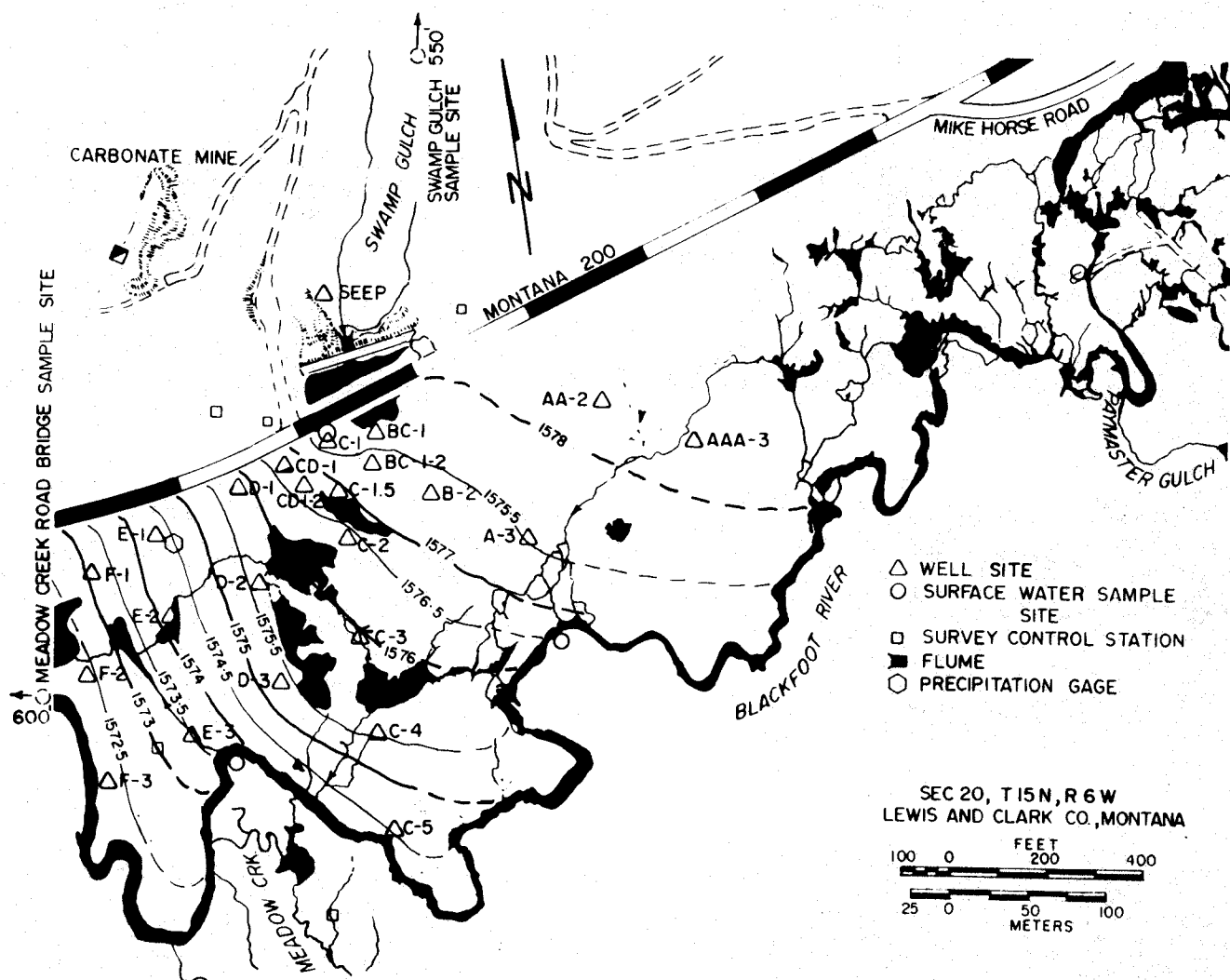


Figure 5-10. Shallow piezometric map on 7/27/87 with 0.5 m contours.

5-4). These values fall within the range of conductivities found in other wetlands studied (Figure 5-4). The range of hydraulic conductivities from the wetland, 1×10^{-2} to 1.5×10^{-5} cm/s, compares to the range commonly found for silty sand (Freeze and Cherry 1979). The decrease in conductivity with depth, excluding the deep wells completed in gravel, is a common occurrence when moving from the acrotelm to the catotelm (Chason and Siegel 1986).

An argument to account for the small conductivity difference between auger holes and wells is that the tests measured part of the same zones. In viewing the depths of auger holes tested

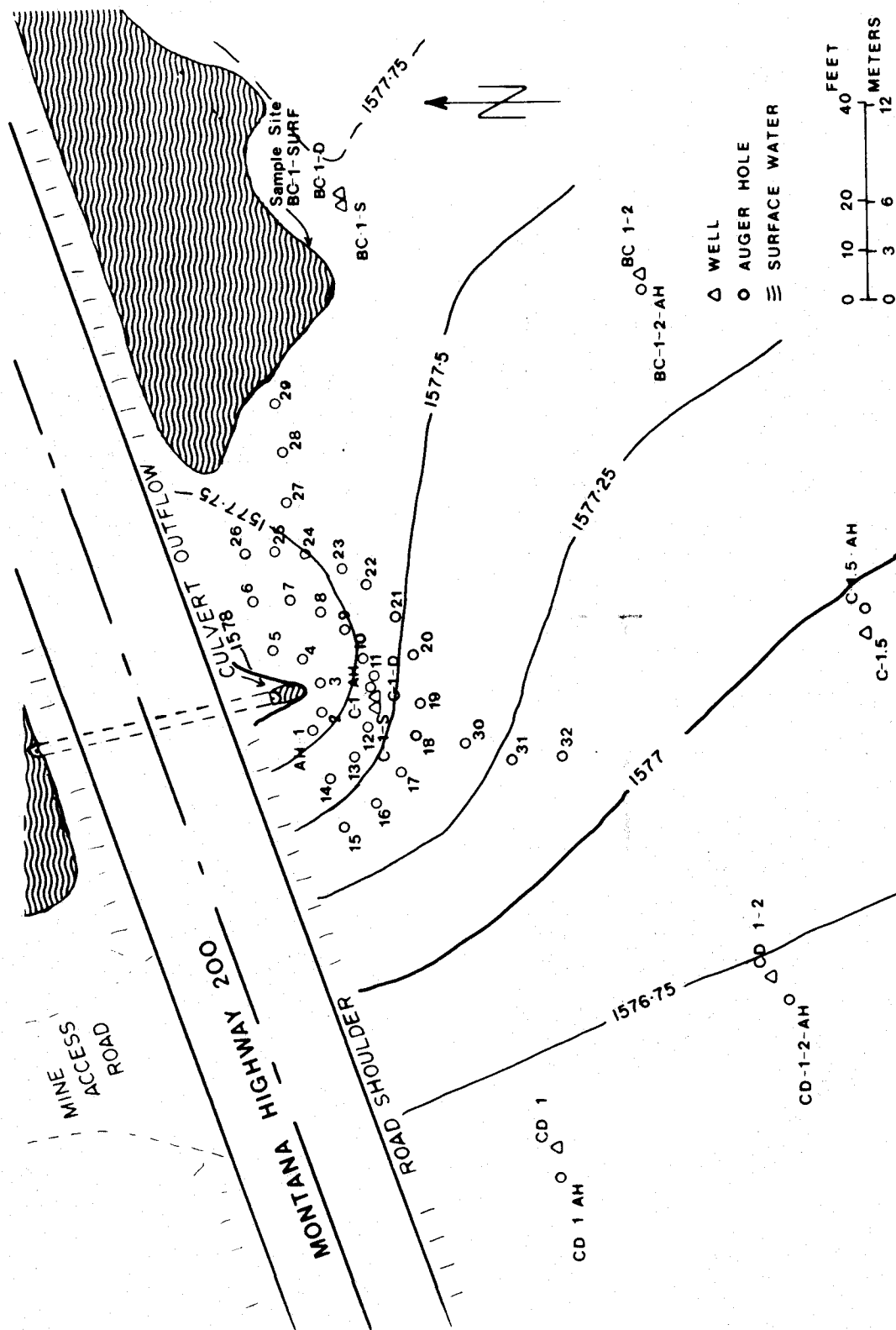


Figure 5-11. Potentiometric surface map on 8/14/87 with 0.025 m contours.

Table 5-3. Field hydraulic conductivity (K), effective porosity (n_e), average hydraulic gradient (dh/dl) and calculated average velocity (V) for each site.

Site	K (cm/s)	n_e (cc/cc)	dh/dl (m/m)	V (cm/s)
B-2-AH	1.06×10^{-3}	.540	.025	4.91×10^{-5}
BC-1-2-AH	5.60×10^{-4}	.540	.025	2.59×10^{-5}
C-1-AH	7.02×10^{-4}	.540	.025	3.25×10^{-5}
C-1.5-AH	1.06×10^{-3}	.540	.025	4.91×10^{-5}
C-2-AH	9.53×10^{-4}	.540	.025	4.41×10^{-5}
CD-1-2-AH	1.18×10^{-3}	.540	.025	5.46×10^{-5}
CD-1-AH	4.54×10^{-4}	.540	.025	2.10×10^{-5}
D-1-AH	4.18×10^{-4}	.540	.025	1.94×10^{-5}
E-1-AH	3.40×10^{-4}	.540	.025	1.57×10^{-5}
E-2-AH	1.08×10^{-3}	.540	.025	5.00×10^{-5}
F-1-AH	1.11×10^{-3}	.540	.025	5.14×10^{-5}
B-2-D	1.01×10^{-2}	.250	.025	1.01×10^{-3}
B-2-S	1.71×10^{-4}	.186	.025	2.30×10^{-5}
BC-1-2	3.89×10^{-5}	.186	.025	5.20×10^{-6}
C-1-D	2.81×10^{-4}	.186	.025	3.78×10^{-5}
C-1-S	6.02×10^{-4}	.186	.025	8.09×10^{-5}
C-1.5	3.00×10^{-4}	.186	.025	4.03×10^{-5}
C-2-D	1.22×10^{-4}	.250	.025	1.22×10^{-5}
C-2-S	1.44×10^{-4}	.186	.025	1.94×10^{-5}
CD-1-2	2.22×10^{-4}	.186	.025	2.98×10^{-5}
CD-1	1.40×10^{-4}	.186	.025	1.88×10^{-5}
D-1-D	1.69×10^{-3}	.250	.025	1.69×10^{-4}
D-1-S	6.95×10^{-5}	.186	.025	9.30×10^{-6}
D-2-D	6.62×10^{-4}	.186	.025	8.90×10^{-5}
D-2-S	1.92×10^{-4}	.186	.025	2.58×10^{-5}
E-1	9.16×10^{-5}	.186	.025	1.23×10^{-5}
E-2	3.21×10^{-5}	.186	.025	4.30×10^{-6}
F-1-D	5.05×10^{-4}	.186	.025	6.79×10^{-5}
F-1-S	1.53×10^{-5}	.186	.025	2.10×10^{-6}

that the auger holes also measured the decomposed organic matter zone (catotelm) layer thereby possibly reducing conductivity. A similar argument is true of the shallow wells some of which had the top of the screened interval and sand pack very near or within the acrotelm. The effect of this situation is that well tests measured a portion of the acrotelm in addition to the catotelm and may have greatly overestimated the actual conductivity of the catotelm. The overall result is an average conductivity from the water table to shallow well total depths, but not the two distinct conductivity differences that might be expected. The average conductivity of both the auger holes and shallow wells is 4.9×10^{-4} cm/s and the velocity is 3.0×10^{-5} cm/s.

Table 5-4. Average hydraulic conductivity (K) from field results and calculated average velocity (V).

Site Type	K (cm/s)	V (cm/s)
Auger Holes	8.11×10^{-4}	3.75×10^{-5}
Shallow Wells	1.68×10^{-4}	2.26×10^{-5}
Deep Wells	2.22×10^{-3}	2.31×10^{-4}
Overall Mean	8.38×10^{-4}	7.14×10^{-5}

The hydraulic conductivity determined in this study is the effect of both vertical and horizontal conductivity. No distinction between vertical and horizontal conductivity was possible with the methods used in the field. It is presumed from other studies (Girts 1986; Chason and Siegel 1986; and Dai and Sparling 1973) that horizontal conductivity would be greater than vertical conductivity in the sedge peat wetland being studied.

Average velocities calculated using field hydraulic conductivities and equation 5-1 are shown in Tables 5-3 and 5-4. Velocities determined were lower than the associated field hydraulic conductivity. This order of magnitude change (Table 5-4) is explained, in part, by the effective porosity chosen for the velocity calculation. From equation 5-1 it can be seen that as effective porosity decreases the water flow velocity will increase if the hydraulic conductivity and gradient remain constant. Another explanation for the lower velocities determined for the field sites measured is the small average hydraulic gradient in the wetland (0.025). The hydraulic gradient of the potentiometric surface in the area of tracer input, culvert outflow to C-1 is 0.058. Incorporating this increased gradient into velocity calculations increases the average velocity from auger hole sites to 8.7×10^{-5} cm/s and velocity from shallow well sites to 5.2×10^{-5} cm/s.

5.3.3 Tracer Experiment

Water flow velocity was also determined in a bromide tracer experiment. Bromide was chosen as the primary tracer in this study for a number of reasons. Many authors have successfully used bromide as a tracer in a variety of environmental conditions (Onken et al. 1977; Smith and Davis 1974; Merrill et al. 1985; and Tennyson and Settergren 1980; to name a few). In addition, Girts (1986) and Bowmer (1987) successfully used bromide to trace wetland water systems. Davis et al. (1985) note that bromide is perhaps the most commonly used ion tracer. Schmotzer et al. (1973) consider bromide to be as close as possible to the ideal tracer. Bromide is a good tracer choice, relative to other

tracers, in the wetland system studied because it is inexpensive, stable, has a low limit of field detection with a specific ion electrode (0.4 mg/L), has low background concentrations, exhibits low toxicity and has low sorption. The relatively few disadvantages of bromide as a tracer are anion exclusion, anion exchange and the presence of other ions which interfere with operation of the electrode. Data from this study indicate bromide is a good choice in the system studied.

Bromide background concentrations and the concentrations of ions which interfere with the electrode were analyzed before the tracer was introduced to the wetland. Background concentrations were analyzed by both the specific ion electrode and by titration (Table 5-5). No auger hole background concentrations were above the 0.4 mg/L limit of detection using the bromide specific electrode. Well water which had background values above the electrode lower limit of detection, 0.4 mg/L, had that value subtracted before plotting the breakthrough curve. The subtraction of background values determined from the bromide electrode was necessary only for five wells. The average bromide value subtracted was 0.47 mg/L with a range from 0.43 to 0.55 mg/L. This subtraction was found to change the shape of the breakthrough curves, but not the placement of the concentration

Table 5-5. Bromide background concentrations in the tracer study area of the Swamp Gulch wetland.

Site	Bromide Concentration (mg/L)	
	Determined by Ion Electrode 9/8 - 9/87	Determined by Titration 7/28 - 31/87
AH-6	< 0.40	---
AH-14	< 0.40	---
AH-17	< 0.40	---
AH-22	< 0.40	---
AH-29	< 0.40	---
AH-32	< 0.40	---
BC-1-2 ¹	0.43	---
BC-1-S	< 0.40	< 0.10
C-1-S ¹	0.44	< 0.10
C-1.5 ¹	0.50	< 0.10
C-2-S	< 0.40	---
CD-1 ¹	0.45	0.10
CD-1-2 ¹	0.55	< 0.10
D-1-S	0.82	0.20
D-2-S	0.40	---
BLANK	< 0.40	< 0.10
BC-1-SURF	---	0.20

¹Backgrounds values subtracted from breakthrough curve.

peak. Bromide background levels measured by titration were never larger than 0.2 mg/L and 80% of all the wetland samples measured were <0.1 mg/L (detection limit). Review of the two methods of analysis suggests that the titrimetric method may be more accurate than the field electrode method, especially at low bromide concentrations. The advantages of using the field millivolt meter and bromide electrode are quick, inexpensive results at the site. This allows for rapid on-site sampling decisions and no need to save a large number of samples for expensive laboratory analysis.

Ions which interfere with the bromide specific electrode, in order of increasing interference, include S^{2-} , CN^- , I^- , NH_3 , Cl^- and OH^- (Orion Research 1982). Sr^{+2} (strontium) must be absent (Cole-Parmer 1987). Water quality analysis (Appendix B) of the acidic wetland system indicate that, of the interfering ions above, only chloride and strontium may cause inaccurate electrode readings. Williams (1979) relates that a chloride concentration of 20 times the bromide concentration will cause error in the bromide determination. At the lower limit of detection with the bromide specific electrode (0.4 mg/L) a chloride concentration of 8 mg/L can cause error. Water analysis at well D-1-S and surface water near BC-1 showed chloride concentrations of 16 mg/L and 8 mg/L respectively on 7/29/87 (Appendix B). The most likely source of this chloride is highway road salt runoff. All wetland sites sampling strontium showed less than the lower limit of detection (0.1 mg/L) except for well F-1-S which showed 0.1 mg/L. Chloride and strontium may thus have created limited variation in the readings for bromide during the tracer test.

The time involved in a uniform (natural) flow tracer method can be quite long and the method is usually limited to short or intermediate distances not more than 100 m (Davis et al. 1985). The velocities determined from the field hydraulic conductivities and preliminary tracer test suggested that the piezometer network might not be close enough to the tracer input point. In addition to variable velocities, conductivity was noticed to decrease with depth and this suggested that flow would be predominantly in the acrotelm. Therefore, a grid of shallow sampling sites was installed nearer the tracer input point. Inexpensive auger holes were used as the sample sites. The preliminary dye tracer test exposed preferential water flow paths and auger holes 27 to 32 were placed along these paths (see Figure 5-6).

Bromide standards ranging from 0.1 to 1000 mg/L were run during each sample session to determine the corresponding mv value for that time period. An example of the resulting calibration curve is shown in Figure 5-12. Regression equations were determined for each calibration curve. The coefficient of determination, (r^2), for each regression was maintained at 0.99. Values which were below the 0.4 mg/L limit of detection were excluded from the regression. Field mv values for each sample

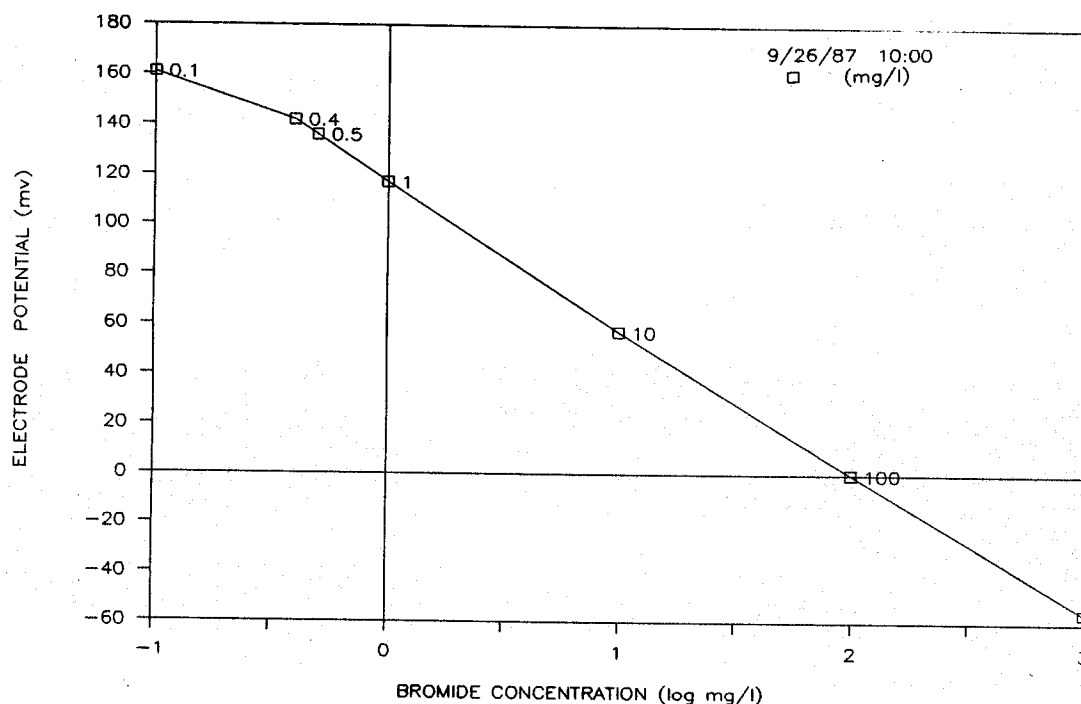


Figure 5-12. Typical calibration curve generated for the tracer study.

site were converted to mg/L values using the regression equation for that corresponding time period. Distilled water blanks were run during each sample session and all were less than the 0.4 mg/L limit of detection.

Bromide breakthrough curves were developed using actual bromide concentrations on the y axis instead of the commonly used concentration/concentration injected (c/c_0). This was done to facilitate calculations and has no effect on the shape of the breakthrough curve. Changing the y axis to concentration/concentration injected is commonly done in tracer studies to facilitate comparison of different types of tracers introduced at the same time. An example of a bromide breakthrough curve generated is presented for site AH-14 in Figure 5-13. Based on the work by Davis et al. (1985) (Figure 5-1), the shape of this curve (steep slopes) indicates little sorption of tracer in the porous medium between the input point and sampling site. The curve shape also exemplifies the expected result that bromide can be considered a fairly conservative tracer in the wetland studied.

There are many possible explanations for shapes of the breakthrough curves. Some of these explanations have been discussed earlier such as retardation by sorption and dilution by dispersion and diffusion. Site AH-25 generated data which

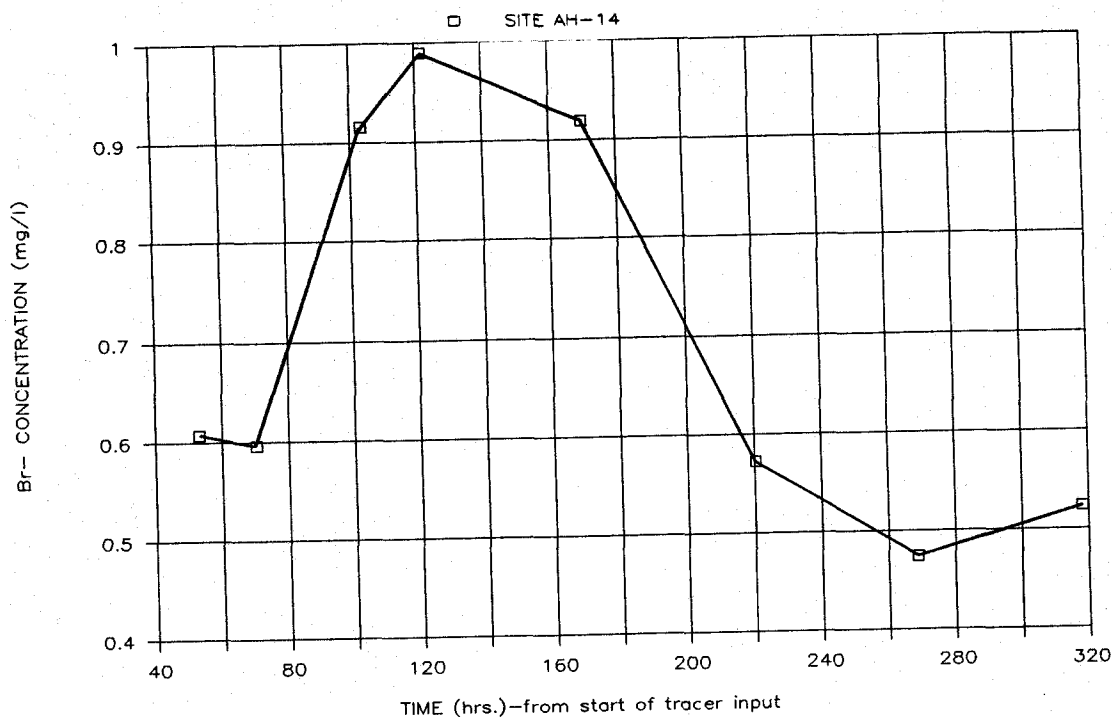


Figure 5-13. Tracer study breakthrough curve for site AH-14, 6.1 m from tracer input showing little effect of sorption.

resulted in only one point above the lower limit of detection. All the neighboring sites, 1.5 to 3.7 m (5 to 12 ft) away generated many bromide breakthrough curve points up to 4.8 mg/L. This suggests zones of very low flow which are described as "dead zones" by Bowmer (1987). At some sample sites the sampling interval was not frequent enough to determine the exact breakthrough peak. In the case of site AH-9 (Figure 5-14) the resultant curve is flat topped between hours 169 and 220. Both of these times were used to calculate velocity in addition to the peaks at 30 and 54 hours. Subtraction of the background values resulted in no detectable tracer peaks on the breakthrough curve for some of the well sampling sites such as C-1-D, D-1-S, and D-2-S.

Placement of the breakthrough curve peak is important in that the associated time represents the passing of the tracer plume. This time is also used for determination of average velocity. Widely spaced sampling intervals may cause the exact breakthrough curve peak to not be found and the velocities determined may be too large or too small. Bromide values below the 0.4 mg/L limit of detection were not considered as possible peaks on the breakthrough curves. A common observation of the

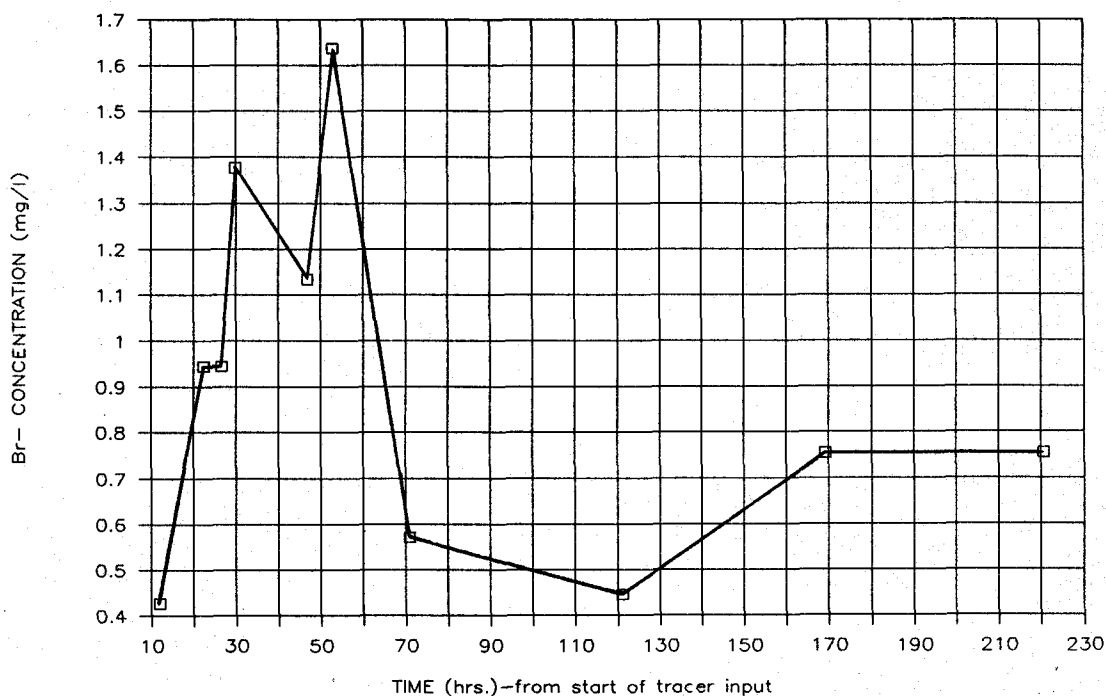


Figure 5-14. Example of infrequent sampling effect on breakthrough curve for site AH-9, 6.1 m from tracer input.

breakthrough curves shows more than one peak of tracer concentration. An example of such a curve is given in Figure 5-15 for site AH-19. This curve seems to indicate two bromide tracer plumes moving past the site with the second plume being more effected by sorption.

How breakthrough curves acquire more than one concentration peak is not specifically discussed in the literature reviewed. Possible explanations of multiple breakthrough peaks could include: inadequate tracer mixing during input, a heterogenous medium with layers of different flow velocity, periodic flushing and/or cross contamination of samples. The inadequate mixing theory is negated by the breakthrough curve shape of the first sampling site encountered (Figure 5-16). This site, AH-3, which showed no multiple peaks and exhibited a high concentration and short time interval to the breakthrough curve peak for this short distance from tracer input. A heterogenous medium is an entirely plausible explanation as layering was indicated during soil sampling (Table 5-2) and preferential flow paths were found in the wetland. These situations are also found in other wetlands (Ingram 1967; Dasberg and Neuman 1977; Sturges 1968). The possibility of samples being drawn from more than one zone of flow is good because many sampling sites have their screened

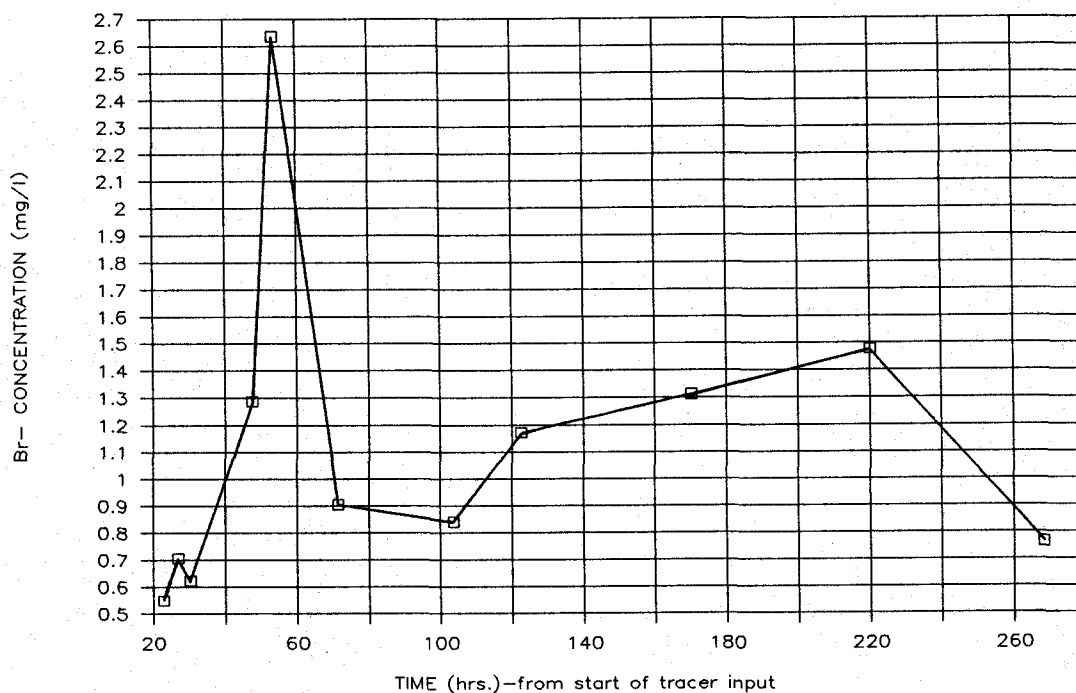


Figure 5-15. Breakthrough curve representing two tracer plumes at site AH-19, 9.1 m from tracer input.

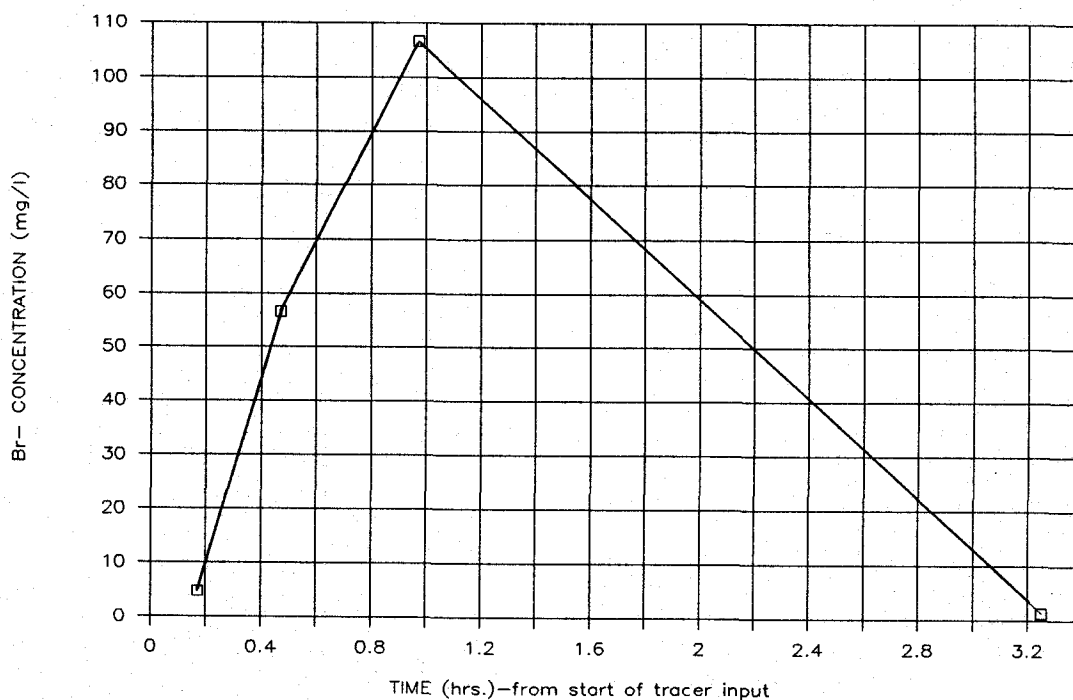


Figure 5-16. Steep breakthrough curve indicating adequate tracer mixing at site AH-3, 3 m from tracer input.

interval in contact with more than one soil horizon (Figure 5-5 and Table 5-2). It is theorized that pulse flow could cause multiple breakthrough peaks by flushing higher concentration water which was previously held in pore spaces during different flow rates. Periodic flushing is indicated by mildly fluctuating water levels during the study period (Appendix B-4). The diurnal variation noticed in flume charts could also contribute to this type of flow.

The possibility of cross contamination of samples, analysis error and/or some other means of acquiring faulty values is one which must be considered in any experiment. Blank samples (distilled water) were run through the same analysis procedures as the wetland samples. The bromide concentration in blanks never was greater than the 0.4 mg/l limit of detection and cross contamination was not a problem. Another explanation for multiple breakthrough curve peaks is error during analysis for bromide in the field. Analysis methods were constant throughout the study and should not have created the noted variation. The final explanation for multiple peaks is a periodic contamination by the interfering ions, chloride and strontium which seems unlikely.

Given these possible explanations for multiple tracer peaks it seems that they are true bromide values and have been processed accordingly. The previously shown breakthrough curve for site AH-14 (Figure 5-13) yields a velocity of 1.4×10^{-3} cm/s (6.1 m/121.7 hr). Table 5-6 shows the velocities associated with the bromide breakthrough curve peak/peaks chosen for each site. The velocities shown represent the rate at which a tracer plume passed that particular site. Site locations are shown in Figure 5-6. The overall mean velocity represented by all of the sites and peaks is 2.6×10^{-2} cm/s with a range of 3.1×10^{-1} to 6.2×10^{-4} .

The velocities were compared in three different ways 1) between auger holes and wells, 2) between the highest peak of the breakthrough curve and the remaining peaks, and 3) between sample sites within the grid area, <18.3 m, and sites outside the grid area, 27 to 38 m (Figure 5-6). These results are shown in Table 5-7. Several observations were apparent: 1) there is little difference between tracer velocities determined at auger holes and wells, 2) the velocities determined from other than the highest peak of the breakthrough curve are notably slower than from the remaining peaks, and 3) there is little difference (same order of magnitude) in velocities determined near the tracer input and those velocities from farther away.

The apparently similar velocities observed at auger holes and wells can be explained by the fact that portions of the

Table 5-6. Tracer velocities determined for each site, time and distance from tracer inputs and peak concentrations of bromide.

Site	Time hrs	Concen- tration mg/l	Distance m	Velocity cm/s	Site	Time hrs	Concen- tration mg/l	Distance m	Velocity cm/s
AH-1	29.6	1.2	3.0	0.00285	AH-28	1.4	20.5	15.2	0.31358
AH-1	53.1	1.6	3.0	0.00160	AH-29	123.1	1.0	18.3	0.00413
AH-2	0.8	3.5	3.0	0.10201	AH-29	268.0	0.4	18.3	0.00190
AH-3	1.0	106.7	3.0	0.08729	AH-30	1.4	52.0	12.2	0.24191
AH-4	29.7	2.6	3.0	0.00285	AH-30	220.2	3.3	12.2	0.00154
AH-4	52.6	1.8	3.0	0.00161	AH-30	317.9	3.9	12.2	0.00107
AH-5	47.1	1.9	3.0	0.00180	AH-31	1.4	42.1	15.2	0.30900
AH-5	52.6	2.2	3.0	0.00161	AH-31	317.9	1.7	15.2	0.00133
AH-6	168.9	1.0	6.1	0.00100	AH-31	411.5	2.1	15.2	0.00103
AH-7	6.2	4.8	6.1	0.02722	AH-32	3.6	29.3	18.3	0.14033
AH-7	52.6	1.8	6.1	0.00322	AH-32	268.6	1.8	18.3	0.00189
AH-8	29.8	3.6	6.1	0.02709	AH-32	411.5	0.7	18.3	0.00123
AH-8	169.1	1.1	6.1	0.00569	C-1-5	52.9	3.8	6.4	0.00336
AH-8	29.8	1.4	6.1	0.00100	C-1-5	121.5	3.1	6.4	0.00146
AH-9	52.7	1.6	6.1	0.00568	C-1-5	220.8	2.5	6.4	0.00081
AH-9	169.1	0.8	6.1	0.00321	C-1-AH	30.0	2.5	6.1	0.00565
AH-9	220.6	0.8	6.1	0.00100	C-1-AH	52.8	1.7	6.1	0.00321
AH-10	52.8	1.1	6.1	0.00077	C-1-AH	121.4	0.7	6.1	0.00139
AH-10	220.6	0.6	6.1	0.00321	C-1-AH	220.7	0.8	6.1	0.00077
AH-11	0.8	82.8	6.1	0.00077	BC-1-SURF	48.6	1.5	27.4	0.01569
AH-11	1.2	82.8	6.1	0.21991	BC-1-SURF	123.7	1.3	27.4	0.00616
AH-11	169.7	0.6	6.1	0.13880	BC-1-S	30.6	1.2	30.8	0.02795
AH-12	0.7	72.9	6.1	0.00100	BC-1-S	54.5	3.0	30.8	0.01568
AH-13	1.5	13.4	6.1	0.24902	BC-1-2	22.6	1.1	35.1	0.04303
AH-13	53.1	9.3	6.1	0.11519	BC-1-2	54.6	2.4	35.1	0.01784
AH-13	169.9	1.9	6.1	0.02776	BC-1-2	915.0	1.0	35.1	0.00106
AH-14	121.7	1.0	6.1	0.00319	BC-1-2-AH	54.6	0.5	34.1	0.01736
AH-15	122.4	1.0	9.1	0.00139	C-1-5	30.7	1.4	37.5	0.03389
AH-16	220.3	0.7	9.1	0.00115	C-1-5	54.8	3.9	37.5	0.01899
AH-16	318.1	0.7	9.1	0.00207	C-1-5-AH	915.2	2.5	37.5	0.00114
AH-17	220.3	1.9	9.1	0.00115	C-1-5-AH	30.7	1.6	37.5	0.03392
AH-18	3.6	36.0	9.1	0.00080	C-1-5-AH	54.9	2.2	37.5	0.01898
AH-18	268.5	4.1	9.1	0.00082	C-1-5-AH	123.9	1.4	37.5	0.00841
AH-19	53.5	2.6	9.1	0.00115	C-1-5-AH	412.5	0.9	37.5	0.00252
AH-19	220.1	1.5	9.1	0.00095	CD-1-2	30.8	1.9	35.7	0.03221
AH-20	53.6	1.6	9.1	0.00474	CD-1-2	54.8	3.5	35.7	0.01808
AH-20	268.4	0.8	9.1	0.00115	CD-1-2	123.9	2.4	35.7	0.00799
AH-21	220.1	1.5	9.1	0.00474	CD-1-2-AH	915.2	2.7	35.7	0.00108
AH-22	53.7	1.5	9.1	0.00095	CD-1-2-AH	54.8	2.9	37.2	0.01884
AH-22	220.0	2.3	9.1	0.00095	CD-1-2-AH	124.0	1.2	37.2	0.00833
AH-23	220.3	4.1	9.1	0.00115	CD-1	269.2	0.8	37.2	0.00384
AH-24	53.8	0.7	9.1	0.00473	CD-1	54.7	3.3	32.9	0.01672
AH-24	268.0	0.7	9.1	0.00115	CD-1	124.0	1.8	32.9	0.00737
AH-25	267.8	0.6	9.1	0.00115	CD-1-AH	915.3	1.3	32.9	0.00100
AH-26	410.7	1.3	9.1	0.00095	CD-1-AH	30.8	1.1	34.7	0.03134
AH-27	53.9	0.9	9.1	0.00095	CD-1-AH	54.7	3.1	34.7	0.01764
AH-27	410.9	1.1	12.2	0.00062	CD-1-AH	124.1	1.3	34.7	0.00778
AH-27	410.9	2.1	12.2	0.00628	CD-1-AH	220.9	0.9	34.7	0.00437
				0.00082	CD-1-AH	412.3	1.0	34.7	0.00234
					CD-1-AH	915.4	0.4	34.7	0.00105

Table 5-7. Tracer velocity comparison.

Comparison Type	Number of Cases	Mean Velocity (cm/s)
Auger Holes	83	2.9×10^{-2}
Wells	18	1.4×10^{-2}
High Peaks	45	5.0×10^{-2}
Remaining Peaks	56	6.4×10^{-3}
Grid Area	70	3.1×10^{-2}
Outside Grid	31	1.4×10^{-2}
Overall	101	2.6×10^{-2}

screened interval of most of the wells are within the same layers of conducting material (Table 5-2 and Figure 5-6) as the auger holes. The total depth of the shallow wells has little effect in that they are able to pick up tracer water from upper layers (acrotelm). This "pick up" can occur through the well sand pack which is usually 30 cm (1 ft) from ground surface, below the bentonite seal. The auger holes are subject to the acrotelm flowing water slightly sooner than the shallow wells as indicated by the slightly faster velocity. The comparison of the high peak and remaining peaks supports the theory of residual tracer water being flushed through after the main tracer plume. These remaining tracer peaks could represent tracer water flowing through the less conductive (catatolem) peat layers hence reducing flow velocity. The higher velocities determined in the auger hole grid area, although within the same order of magnitude, may reflect the steeper gradient in the area near the culvert outflow. Considering all the velocities from the different comparisons it is concluded that the overall mean of 2.6×10^{-2} cm/s is the most representative of the wetland system measured.

The presence of preferential flow paths is demonstrated by results of the bromide tracer experiment. Breakthrough curves peak at just 1.4 hours from tracer input for sites AH-28 and AH-31, both 15.2 m (50 ft) from tracer input (Table 5-6). This results in a tracer plume of the shape represented in Figure 5-17. These preferential flow paths indicate the tracer is moving in both a south-southwesterly and easterly direction. The bromide concentration at site AH-31 is more than twice as high (52 mg/L) as site AH-28 (20.5 mg/L) at the same time (1.4 hrs) and indicates that more tracer is flowing in the south-southwesterly direction. This southwesterly skewed radial pattern is expected as the culvert outflow is a topographic high point and

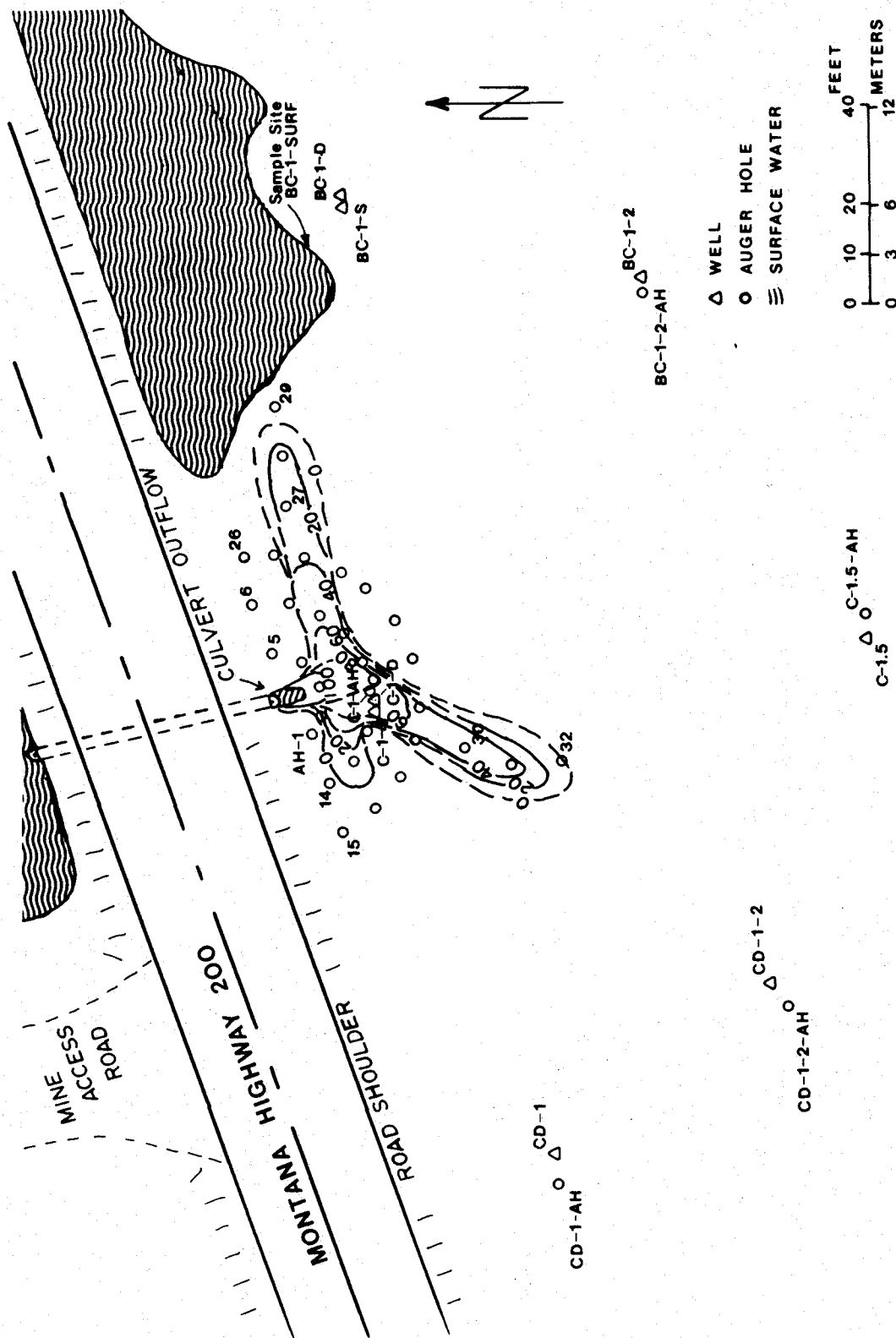


Figure 5-17. Bromide concentration isopach map, 0.95 to 1.5 hours from tracer start (20 mg/L contours).

the regional hydraulic gradient and topographic slope is to the southwest. Preferential flow paths are found in man made wetlands (Bowmer 1987) and in natural systems are sometimes called water tracks (Ingram 1967). Such paths may cause a substantial proportion of the water (tracer and/or AMD polluted) to travel through the system faster than predicted by the field hydraulic conductivities. Brooks (1984) recommends man made wetlands be designed to avoid channelization (preferential flow paths) and encourage low velocity sheet flow by use of flow obstructions. Burris et al. (1984) shows that as flow rate decreases effective iron removal from AMD increases in a wetland.

Tracer flow velocities were faster than flow velocities estimated from field hydraulic conductivity tests on auger holes and wells. The relationship between tracer determined velocity and field hydraulic conductivity determined velocity is complicated by a number of factors. The measurement of the conductivities and the tracer test were done at different times. In effect, the measures are of different systems, but in practice, are assumed the same in view of the minimal change noticed in the system over an extended period. The field conductivity determined velocities are more representative of the wetland at depth. The tracer velocities, which include preferential flow paths, are more representative of the true movement of AMD waters. The amount of area and time represented by the tracer test is large compared to the small area and time that the field conductivities represent. This infers that the tracer velocities are closer to the true value than the conductivity determined values.

5.4 SUMMARY

Bromide was a good choice for tracing water movement in the wetland studied. Using the natural flow tracer technique, sufficient bromide concentrations were detected up to a distance of approximately 38 m from tracer input. The bromide specific ion electrode method of analysis was found to be adequate. This method could be improved by periodic double sampling and analysis with another method (titration) especially at low bromide concentrations.

The wide range of velocities determined from this study are the effect of a very heterogeneous wetland flow system and the methodology used in determining the velocities. Some of the heterogeneity involved a change in hydraulic gradient, changing flow rates and periodic flushing over time. Preferential flow paths and "dead zones", the result of variable hydraulic conductivity, also add to the heterogeneity. The most representative velocity of the AMD water in the wetland was that

determined by the tracer experiment. These velocities were found

to be a reflection of the heterogeneous system and ranged between 3.1×10^{-1} to 6.2×10^{-4} cm/s with a mean of 2.6×10^{-2} cm/s (73.7 ft/day).

Comparison of field determined hydraulic conductivity, from tests of auger holes and wells, to tracer determined velocity was accomplished by converting conductivity to velocity utilizing the estimated values of effective porosity and hydraulic gradient. These field conductivity determined velocities are reasonable in that they estimate water flow velocity in the small area where conductivity was determined. When comparing the conductivity determined velocity to tracer velocity even the average velocity of auger hole sites (fastest of the conductivity determined velocities) was three orders of magnitude lower than average tracer velocity. This order of magnitude relationship does not change even if effective porosity is changed to 0.1 (low) and the hydraulic gradient is changed to the steepest observed in the study area (0.058 m/m). This relationship infers the comparison is that of two different flow systems. The tracer velocity represents preferential flow paths within the shallow (acrotelm) system and the conductivity represents that flow found at greater depth (catotelm).

The determination of water residence times in the wetland is contingent upon the velocity of the water system to be considered and the distance the water travels. The concern in this study involves the AMD water which emanates from the culvert outflow into the wetland. The velocity representing this source is that of the tracer which was input at the same point that the AMD water is introduced to the wetland. The distance the AMD water travels, before complete problem amelioration, was not determinable from the tracer methods employed, but is estimated to be approximately 200m as observed in acrotelm soil iron concentrations (Appendix C and Section 8). The shallow groundwater iron concentrations show a trend similar to the soils by decreasing in the down-gradient direction and approaching background iron concentrations near site F-1. With the 200m distance chosen and using the mean tracer velocity, the average residence time of AMD water in contact with the wetland acrotelm soils is estimated to be 215 hours (9 days). This estimate is considered to be quite variable as noted earlier and represents a low flow regime (Autumn). This residence time estimate should be viewed with caution as it pertains only to iron amelioration and the velocity used represents only that measured at the time of the tracer study for the very limited area of the wetland near the AMD discharge point. Assuming the wetland studied is a viable system for the amelioration of AMD, the challenge in the design of man made wetland systems will be to determine how to emplace similar hydrologic characteristics, and limit preferential flow paths.

6.0 WETLAND INFLUENCE ON MINE DRAINAGE WATER QUALITY

The water quality sampling program for the Swamp Gulch study was initiated to determine: 1) the concentrations of parameters in the AMD; 2) the background water quality in Swamp Gulch above the Carbonate Mine and on the Blackfoot River above the wetland; 3) the changes in AMD water quality as it progressed through the wetland and; 4) changes in the water quality of the Blackfoot River below the wetland discharge point.

6.1 SAMPLE COLLECTION AND PROCESSING PROCEDURES

Water samples were collected by various means, each appropriate to the sample source. All samples were initially collected in 4 liter (1 gal) linear polyethylene (LPE) (Nalgene) decontaminated bottles (acid/deionized water cleaned). These bottles were rinsed three times with sample water prior to filling. The filled 4 liter bottles were placed in coolers with ice until divided into appropriate aliquots for preservation, usually within minutes after collection.

Water level depths were recorded at all wells prior to pumping. These measurements were referred to the top of the well casing (as surveyed) and recorded to the nearest 0.01 foot using a Soil Test Model 444-000 or Johnson UOP water level indicator.

Well samples were acquired with a peristaltic pump (Masterflex) using a length of silicone tube. The apparatus was rinsed with distilled or deionized water after each sample, and if organic deposition on the hose was evident, new hose was installed. A minimum of three well volumes (including sand pack volume) of water was pumped before sampling at the higher yield sites. At low yield sites, wells were pumped dry and the recharge was sampled.

Surface water samples were obtained by submersing the four liter bottles below the water surface where sufficient water depth was present or by filling the four liter bottles with a peristaltic pump where small volumes of water required this method. Flow measurements or estimates were made at the time of sample collection for all flowing surface water sites.

Various aliquot portions were taken from the four liter bottles for the required analyses. The samples were preserved in three separate bottles: 1) 125 or 250 ml of 0.45 micron filtered sample with 1 ml of concentrated HNO_3 preservative (metal analyses), 2) 125 or 250 ml of 0.45 micron filtered sample with one ml of concentrated H_2SO_4 preservative (NO_3 analysis) and 3) 250 or 500 ml of nonfiltered sample with no

added preservative (pH, electrical conductivity (EC), major cations, HCO_3 , F, P, Cl, SO_4 analyses). Special bottles were prepared for the limited number of dissolved organic carbon and cyanide analyses.

Samples were filtered utilizing a Horizon Ecology peristaltic pump (silicone hose) and a Geotech 102 mm filter apparatus. The pump and filter assembly were rinsed thoroughly with deionized water between samples. Filters and prefilters were changed between all samples except field replicates. All sample bottles were rinsed three times with sample water (filtered or nonfiltered according to aliquot) prior to filling. A minimum of one field blank was included with each sample collection and inserted blind within the sample train. The blank was processed in a manner identical to that of the normal samples. Blind field replicates were processed from the same four liter initial sample. Blind EPA field standards were processed in the same manner as were field blanks. All processed samples were immediately placed in coolers with ice or refrigerated.

Temperature, conductivity (EC) and pH measurements were made (in-situ where possible) at the time of sample collection or, under adverse conditions (ambient temperatures below freezing), they were determined within one to three hours in a sheltered environment. The temperature was determined with a mercury thermometer, dial thermometer or a Cole-Parmer SL-5985-80 pH meter with a digital temperature readout. These instruments were evaluated with an ice bath and a laboratory mercury thermometer for accuracy and, in all cases, were accurate within 1°C . Sample pH was determined using either a Cole-Parmer Model 5986-00 or a Cole Parmer Model SL-5985-80 pH meter. The latter instrument was rated to maintain calibration up to one year. It was calibrated at the beginning of each sample trip and checked daily. The instrument was recalibrated if the pH varied more than 0.2 pH units from standards. The Model 5986-00 pH meter was standardized for each sample.

The EC of samples was measured with a Lamotte Chemical Model DA-1 Conductivity Bridge or a Cole-Parmer Model R-1481-50 Conductivity Meter. The Lamotte unit was standardized for each analysis. The Cole-Parmer unit was standardized at the beginning of each sample trip and checked twice daily thereafter. This unit is rated to maintain calibration up to 90 days.

All data were recorded in the project field book or on site matrix sheets.

At the completion of sample collection and processing, samples were stored under security until shipped by bus to Energy Laboratories in Billings, Montana. Although strict EPA custody procedures were not followed due to the expense, at no time were samples left out of sight, unattended or unsecured.

The water sampling program for the Swamp Gulch wetland study began with initial sampling during the April 29 - May 2, 1987 period, following completion of the project installation. The largest surface flows through the wetland were observed at this time, but the flume on Swamp Gulch had not yet been installed so only a one point estimate of Swamp Gulch flow was noted (estimated at 20 to 30 gpm). The second set of samples was collected from July 28 to July 31, 1987, and included samples from piezometers F-1-S and F-1-D which were installed on July 28, 1987. The third set of samples, collected during the January 11 to January 13, 1988 period, consisted of a partial set confined largely to the area of estimated maximum acid drainage impact with several appropriate background sites. Low flow conditions prevailed during both the second and third sampling trips.

6.2 ANALYSES METHODS

All analyses were made utilizing standard EPA procedures (Appendix A). These are detailed in Methods for Chemical Analysis of Water and Wastes (EPA 1983).

6.3 QUALITY CONTROL/QUALITY ASSURANCE

The water quality program was designed to evaluate the validity of these data. Field sampling included blind field replicates, blind field water blanks and blind field standards. Laboratory procedures included duplicates and spikes. These data have been utilized to generate accuracy and precision statements following standard procedures for evaluating data from the EPA Contract Laboratories Program (Appendix A). These data are summarized in the following sections.

6.3.1 Accuracy

An evaluation of five Environmental Protection Agency (EPA) standards submitted with samples indicate laboratory analyses somewhat overestimated Ca, Mg, SO_4 , HCO_3 , TDS and NO_3 (Table 6-1). The laboratory did not recover any phosphate as orthophosphate from standards containing 0.50 and 0.27 mg/L phosphate. This problem is the result of running phosphate analyses on the unpreserved aliquot which did not contain the PO_4 standard. The laboratory spikes for PO_4 suggest an adequate recovery (96 percent) for this parameter. Three EPA standards were submitted for trace element analyses. These data generally exhibit low standard deviation values and little bias (Table 6-2). The variation between the number of samples was due to the changing of the suite of analyses to be performed as the project continued.

Table 6-1. Accuracy results based on blind field standards.

Parameter	Mean % Recovery	Range of Uncertainty	Standard Deviation	Number of Samples	Bias Correction	Completeness(%)
Ca	108.0	+ 7.6	7.97	5	x0.93	100
Mg	106.6	+ 5.7	6.02	5	x0.94	100
Na	103.2	+ 4.9	5.17	5	None	100
K	97.0	+ 2.6	2.74	5	x1.03	100
SO ₄	111.4	+ 7.8	8.17	5	x0.90	100
HCO ₃	117.2	+21.9	22.95	5	x0.85	100
Cl	101.2	+ 4.6	4.82	5	None	100
F	97.8	+12.4	13.05	5	None	100
P	----	----	----	0	----	----
NO ₃	122.0	+11.6	9.83	4	x0.82	100
TDS	119.4	+19.2	20.17	5	x0.84	100
EC	101.0	+ 1.9	2.00	5	None	100
PH	93.2	+11.3	11.82	5	None	100
Al	97.3	+ 3.9	2.31	3	None	100
Sb	----	----	----	0	----	----
As	97.0	+ 6.3	1.41	2	None	100
Cd	101.3	+10.3	6.11	3	None	100
Cr	100.0	----	----	1	----	----
Co	95.0	+31.6	7.07	2	None	100
Cu	98.5	+ 9.5	2.12	2	None	100
Fe	102.3	+11.2	6.66	3	None	100
Pb	99.0	+12.6	2.83	2	None	100
Mn	97.7	+ 4.2	2.52	3	None	100
Ni	100.5	+ 3.2	0.71	2	None	100
Se	----	----	----	0	----	----
Ag	----	----	----	0	----	----
Zn	99.0	+ 6.3	1.41	2	None	100

Table 6-2. Accuracy results based on laboratory spikes.

Parameter	Mean & Recovery	Range of Uncertainty	Standard Deviation	Number of Samples	Bias Correction	Completeness(%)
Ca	103.6	+4.6	4.83	5	None	45
Mg	103.2	+4.3	4.49	5	None	45
Na	102.4	+4.5	4.72	5	None	45
K	109.6	+3.9	4.04	5	x0.91	45
SO ₄	97.8	+2.3	2.39	5	None	45
HCO ₃	Not Run					
Cl	99.4	+5.3	5.59	5	None	45
F	101.6	+2.6	4.70	11	None	100
P	96.2	+3.0	5.47	11	x1.04	100
NO ₃	99.2	+2.0	3.63	11	None	100
TDS	Not Run					
EC	Not Run					
pH	Not Run					
Al	103.1	+3.3	6.0	11	None	100
Sb	98.7	+5.3	6.44	6	None	100
As	100.2	+8.5	8.93	5	None	100
Cd	100.9	+1.7	3.11	11	None	100
Cr	108.5	+0.7	0.58	4	x0.92	100
Co	98.4	+2.8	2.97	5	None	100
Cu	102.0	+2.2	2.61	6	None	100
Fe	100.0	+1.9	3.44	11	None	100
Pb	96.0	+3.6	4.83	7	x1.04	100
Mn	98.8	+0.8	1.47	11	x1.01	100
Ni	99.8	+1.7	2.04	6	None	100
Se	106.5	+2.0	1.73	4	x0.94	100
Ag	88.5	+6.8	8.29	6	x1.13	100
Zn	100.1	+4.5	7.82	11	None	100

1 At 90% Confidence Level

6.3.2 Precision

Precision calculations were based on blind field replicates and laboratory duplicates. The maximum range of uncertainty for laboratory duplicates (at the 90% confidence level) was 7.69% for HCO_3 (Table 6-3). Precision calculations based on blind field replicates generally indicated good precision (at the 90% confidence level) except for sodium, fluoride, manganese and total suspended solids (TSS) (Table 6-4). The high imprecision for sodium is the result of values at or near the instrument detection limit where a 1 to 2 ppm difference in analyses produced large relative standard deviations. High uncertainty values for fluoride and manganese were due to the small number of valid samples (2 each) and the resulting high t values. The precision of total suspended solids (TSS) analyses based on blind field replicates was strongly influence by one sample (from a total of 2). The TSS sample and replicate sampled on April 29, 1987, exhibited values of 16 and 8 ppm, respectively, for this parameter which resulted in a 67 percent relative standard deviation and a precision value of ± 298 percent.

6.3.3 Cross Contamination

Evaluation of the field water blanks indicated minimal cross contamination (Table 6-5). No chemical parameters were above the instrument detection limit except bicarbonate for the April 1987 and January 1988 sampling trips. Some bicarbonate is usually present in all blanks due to carbon dioxide present in the atmosphere, a portion of which dissolves in water forming bicarbonate. One out of the two blanks for the July 1987 trip exhibited a sulfate value of 40 mg/L, likely as the result of sulfuric acid (H_2SO_4) used for preservation. The bottles may have been mislabeled or the laboratory may have run sulfate on the wrong aliquot. It is not felt that an actual contamination by sulfate occurred as all other parameters associated with this blank, with the exception of TDS, were below the detection limits. Sulfate values associated with normal water samples for the July 1987 sample trip do not show any associated jump in concentration.

6.4 SURFACE WATER QUALITY

6.4.1 Swamp Gulch Above the Carbonate Mine

The water quality of Swamp Gulch above the mine is good and meets most criteria for drinking water (USPHS 1962; EPA 1978, 1979), Livestock (NAS 1974), aquatic life (EPA 1983; NAS 1974) and irrigation (NAS 1974). The only exception was an enormously high (0.21 mg/L) manganese value obtained during the January 1988 sample collection. The April and July, 1987 values for manganese were both below the 0.02 mg/L instrument detection

Table 6-3. Precision, based on laboratory duplicates, at the 90% confidence level.

Parameter	Precision	Completeness	Comments
Ca	+ 3.2%	36%	Eliminating one sample set near Instrument Detection Limit (IDL)(1 and 2 mg/L for sample and duplicate respectively), precision= <u>12.3%</u>
Mg	+ 4.0%	45%	
Na	+26%	45%	
K	+ 0.0%	27%	One sample above IDL, no duplicate Two samples, both IDL Eleven samples, all IDL
SO ₄	+ 0.3%	45%	
HCO ₃	+ 7.6%	27%	
Cl	+ 0.0%	45%	
F	+ 1.7%	91%	
Br	-----	-----	
I	-----	-----	
P	-----	-----	Eleven samples, all IDL
NO ₃	+ 2.3%	100%	
Al	+ 1.2%	55%	
As	-----	-----	
Cd	+ 4.5%	55%	One sample IDL, RPD=0.0%
Co	-----	13%	
Cu	+ 0.0%	38%	Five samples, all IDL
Cr	-----	-----	
Fe	+ 0.4%	82%	One sample IDL, RPD=0.0%
Pb	+ 0.0%	50%	
Mn	+ 0.4%	100%	Eight samples, All IDL
Ni	+ 0.0%	25%	
Se	-----	20%	One sample IDL, RPD=0.0%
Ag	-----	-----	
Sb	-----	13%	One sample IDL, RPD=0.0%
Sr	-----	-----	
Zn	+ 0.9%	90%	One sample, RPD=0.0%
pH	+ 0.0%	45%	
EC	+ 0.8%	45%	One sample, IDL
TDS	+ 5.0%	18%	
TSS	+ 3.2%	45%	One sample, IDL
DOC	-----	100%	
CN	-----	-----	

Table 6-4. Precision, based on field replicates, at the 90% confidence level.

Parameter	Precision	Completeness	Comments
Ca	± 0.0%	100%	-
Mg	± 4.4%	100%	-
Na	±46.0%	100%	-
K	± 0.0%	50%	-
SO ₄	± 1.2%	75%	-
HCO ₃	± 6.0%	100%	-
Cl	± 0.0%	50%	-
F	±81.0%	50%	-
P	NC	0%	All values <IDL
NO ₃	±13.6%	75%	-
Al	NC	0%	All values <IDL
Cd	NC	0%	All values <IDL
Fe	5.6%	75%	-
Pb	NC	0%	All values <IDL
Mn	27%	50%	-
pH	2.2%	100%	-
EC	2.5%	100%	-
TDS	8.0%	100%	-
TSS	298.0%	67%	-

NC = Not calculated

limit. The high sample value may have been due to the amount of sediment disturbed during sampling (ice removal). Swamp Gulch water above the mine has exhibited little variation during the period of study and is characterized by a calcium bicarbonate water type. Calcium, magnesium, bicarbonate and sulfate percentages in equivalents/L are 54.6, 37.9, 80.4 and 19.6, respectively. Slight increases were noted for both sodium (2 mg/L increase) and sulfate (7 mg/L increase) during the January 1988 sample period and are of little significance.

6.4.2 Swamp Gulch Below the Carbonate Mine

Swamp Gulch water below the mine consistently exceeded drinking water criteria (USPHS 1962, EPA 1978, EPA 1983) for aluminum, cadmium, iron and manganese. The water was characterized by an iron sulfate water type in which iron and sulfate comprised 31.1 and 99+ percent of cations and anions, respectively, assuming there was an even balance between ferrous and ferric iron forms. Other major cations were calcium (27.6 percent), magnesium (18.7 percent), aluminum (16.9 percent),

Table 6-5. Field Blank.

Date	1/12/88	7/29/87	4/29/87	7/31/87
Sample Designation	AAA-4	Surf-12	Surf-3	Surf-14
K	<1	<1	<1	<1
Na	1	<1	<1	<1
Ca	1	<1	<1	<1
Mg	<1	<1	<1	<1
SO ₄	1	<1	<1	40
Cl	<1	<1	<1	1
HCO ₃	2	2	2	2
TDS	1	18	<1	13
TSS	--		4	
EC	1	2	1	2
PH	5.0	5.1	5.0	5.0
F	<0.10	<0.10	<0.10	<0.10
P	<0.01	<0.01	<0.01	<0.01
NO ₃	<0.05	0.21	0.15	0.19
Al	<0.1	<0.1	<0.1	<0.1
Cd	<0.001	<0.001	<0.001	<0.001
Fe	<0.03	<0.03	<0.03	<0.03
Pb	<0.01	<0.01	Not Run	<0.01
Mn	<0.02	<0.02	<0.02	<0.02
As		<0.005	<0.005	
Co		<0.01	<0.01	<0.01
Cu		<0.01	<0.01	<0.01
Ni		<0.03	<0.03	<0.03
Sb		<0.01	<0.01	<0.01
Ag		<0.005	<0.005	<0.005
Cr		<0.02	<0.02	
Se		<0.005	<0.005	
Zn		<0.03	<0.01	<0.01
Sr		<0.1	<0.1	<0.1

manganese (3.3 percent) and sodium (2.3 percent). This water also exhibited an increase in sulfate and iron in the January 1988 sampling. The increase in iron was especially notable (from 18.8 to 48.4 mg/L) and likely represents the combined result of several factors: 1) low flow which has less dilution; 2) the presence of an ice cover, which may have restricted oxygen availability (and subsequent precipitation of iron as iron-hydroxide) and, 3) somewhat lower pH which would allow more iron in solution. This observation suggests physio-chemical precipitation of iron may be important in this system and that this process starts before the AMD is discharged into the wetland.

6.4.3 Blackfoot River

The general water type of the Blackfoot River at Pass Creek, located upstream of the wetland study area, was consistent throughout all three sample periods (Figure 3-1). The dominant water type, expressed in milliequivalent per liter, was a calcium bicarbonate in which calcium and bicarbonate composed 53 percent of both cations and anions respectively. Magnesium and sulfate represented 44 and 47 percent of cations and anions. Little change occurred in these percentages during high or low flow. The water type for the Blackfoot River at the Meadow Creek road bridge, located downstream of the wetland study area, exhibited a marked change during low flow. The April and July 1987 water samples exhibited a similar cation-anion balance to that found at the Pass Creek site (54, 43, 52 and 48 percent for calcium, magnesium, bicarbonate and sulfate, respectively). However, during the January 1988 sample period, a slight decrease in calcium (to 51 percent), an increase in sodium (from 2.1 to 4.6 percent) and a gross increase in sulfate (69.7 percent) at the expense of bicarbonate (30.3 percent) was evident. The ramifications of this will be discussed later.

6.4.4 Background Statistical Comparisons

Two general comparisons have been evaluated: 1) the Swamp Gulch water quality above and below the Carbonate Mine and 2) the Blackfoot River above and below the wetland discharge area. The Swamp Gulch drainage exhibited significant to very highly significant increases in the concentrations of SO_4 , Al, Mn, Zn, Cd, Fe, Pb and Cu when compared using a paired "t" test. No significant increases were observed for As, Cu, Co, Sb, Se and Ag at the given detection limits (Appendix B-3). The "t" test comparison of the Blackfoot River at Pass Creek and at the Meadow Creek Road Bridge indicated no significant differences. However, the limited number of samples (3) for each site did suggest the Meadow Creek Road Bridge sample site was slightly higher in iron, manganese and sulfate. The slight increases in parameters in Blackfoot River samples below the Swamp Gulch wetland discharge point as opposed to those above the wetland

are great enough to account for approximately one half of the iron and all of the manganese contributed by the Swamp Gulch acid mine drainage. Zinc is the only other trace metal present in the Blackfoot River at notable levels and was apparently diluted in the river reach adjacent to the wetland. Both the April and July sample sets show the highest zinc levels in the upper two river sample sites with each downstream sample site showing progressively lower zinc concentrations. This effect could not be determined during January 1988 due to the reduced number of samples taken at that time. These data strongly suggest the source of the zinc is either Paymaster Gulch, the main fork of the Blackfoot River immediately above the study area or some discharge directly into the river or wetland. The latter may be some flowing abandoned drill holes, one of which was measured for EC and pH (352 umhos/cm and 3.97, respectively) on April 29, 1987.

6.4.5 Wetland Surface Water

In addition to the Blackfoot River and Swamp Gulch samples, surface water samples were obtained from six surface sites within the wetland during the July sample trip. These were obtained at sites AAA-3, BC-1, C-1.5, C-3, CD-1 and F-1. Water quality parameters at site AAA-3 were very similar to the Blackfoot River at Pass Creek samples and exhibited a calcium bicarbonate water type and low levels of all metals (iron at 0.07 mg/L). This would be expected from the observed drainage system. Parameters at site C-3 were also similar to the river samples but iron and manganese levels were elevated to 6.94 and 0.24 mg/L, respectively. This suggests some wetland drainage is mixing with the diverted river water at this point. The pond sample at BC-1 was intermediate in composition between values observed at C-3 and the Swamp Gulch drainage which suggests this pond is being partially recharged from the upgradient, non-polluted wetland area. The water quality at sites C-1.5, CD-1 and F-1 was similar to the Swamp Gulch AMD discharge. The water type at C-1.5 was an iron sulfate and at sites CD-1 and F-1 it was a calcium sulfate. Sulfate was the only major anion in the latter samples. High levels of sulfate, iron and manganese would be expected at sites C-1.5 and CD-1, located 37.5m (123 ft) and 32.9m (108 ft), respectively, from the Swamp Gulch discharge point. However, the high levels observed at site F-1 raise serious questions as to the actual effectiveness of this wetland in ameliorating AMD water. The process may be one of partial dilution which cannot be accurately accessed with the present instrumentation. It should also be noted that these conclusions are based on one set of surface samples which could be subject to analytical or sampling errors which are not apparent.

6.5 GROUND WATER QUALITY

The ground water of the wetland research area has been characterized by samples from two zones, shallow and deep. The former is generally from 0.6 to 1.2 m (2 to 4 ft) and the latter varied up to a maximum of 4.9 m (16 ft). The shallow zone screened interval generally spanned portions of both the surficial organic layer (acrotelm) and the underlying muck layer (catotelm). The deep wells bottomed in the gravel underlying the muck layer. However, extreme difficulty was encountered trying to keep the deep bore holes open long enough to place the screen within the gravel zone and as a consequence, most deep wells have a significant portion of their screened interval within the muck zone. All wells were sandpacked and, on the deep wells, the sand pack extended into the partially caved gravel portion of the bores. The water quality data strongly suggest most water withdrawn from the deep wells is from the gravel aquifer. The following sections describe water quality parameters of the shallow and deep zones. An evaluation of the two wells in the acid drainage seep have been evaluated separately in section 6.5.3.

6.5.1 Shallow Ground-water System

The shallow ground-water system (0-1.2m, 0-4ft) is dominated by iron-calcium sulfate and calcium-magnesium bicarbonate water types. The former is present throughout the main Swamp Gulch AMD impact area, the latter is generally confined to the southwestern portions of the study area (Figure 6-1). A calcium-magnesium sulfate water type is conspicuously present at two sites near the Blackfoot River (C-5 and F-3). The only other site at which the calcium-magnesium sulfate water type was dominant is the site AA-2. It is noteworthy that background site AAA-3 exhibits the same water type found in the AMD impact area although overall concentrations are lower at site AAA-3. The zone dominated by the bicarbonate anion is confined to the area which apparently receives water from the Blackfoot River via channels flowing past sites C-3 and E-3. The single sample from site F-1 was the only analyses in which bicarbonate was the dominant anion and which was not affected by Blackfoot River water.

A plot of mean dissolved iron values for all well sites clearly indicates that the shallow ground-water system is enriched in iron as the result of AMD (Figure 6-2). Background means for iron ranged from <0.03 to 27 mg/L with the very low values (<0.03 and 0.25 mg/L) associated with sites E-3 and F-3, respectively. Sites E-3 and F-3 are apparently recharged directly from the Blackfoot River. Samples from shallow well sites near AMD discharge point are 2 to 3 times above both the background shallow ground-water concentrations and the inflow AMD concentrations. The highest mean (73 mg/L) was found for

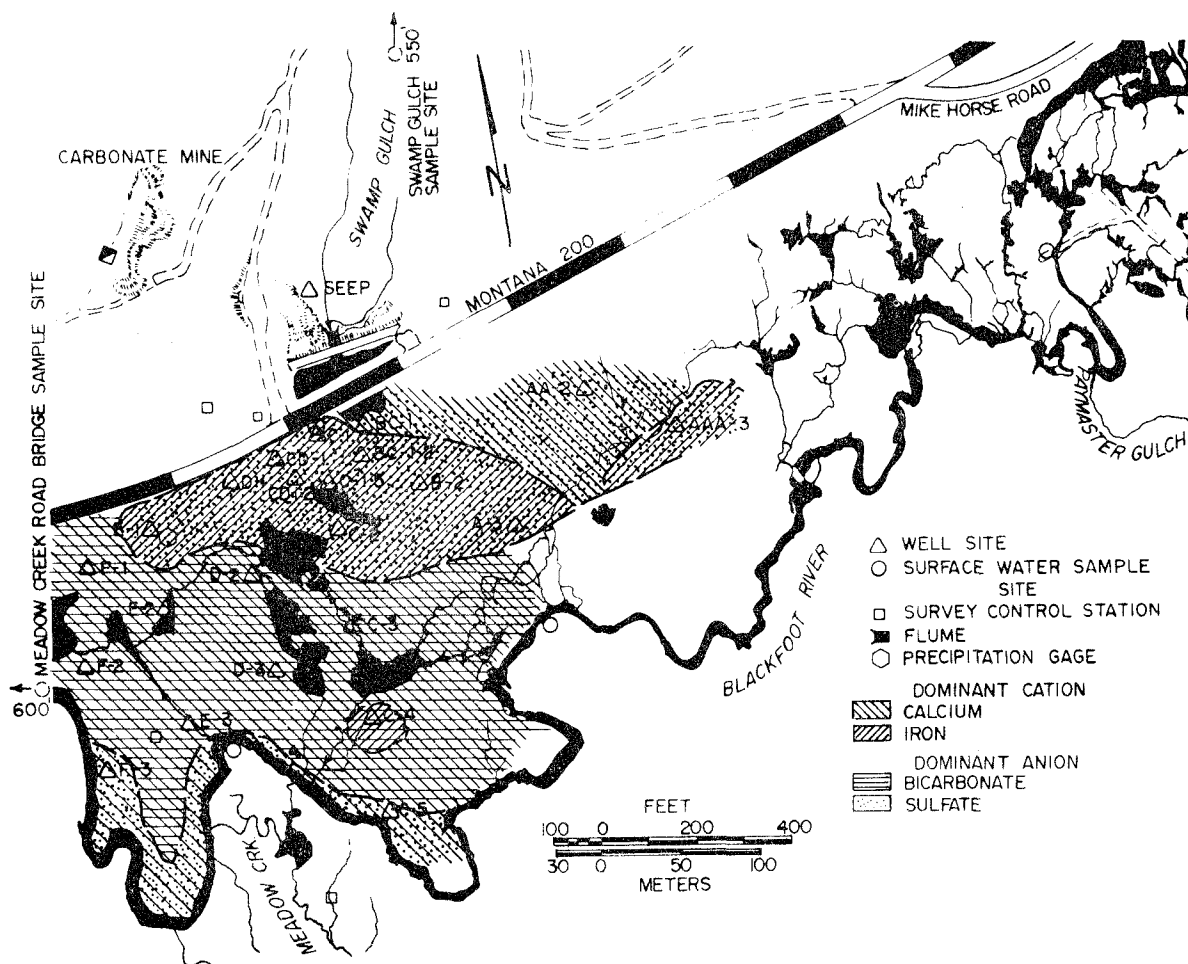


Figure 6-1. Shallow ground-water types for the wetland system.

site C-1, the closest site to the AMD discharge point. Dissolved iron concentrations fall abruptly eastward towards site B-2, at least partially the result of lower ground-water gradients in this direction. Concentrations fall more gently towards the southwest, which suggests more AMD flows in this direction which is consistent with topography and piezometric maps. Iron concentrations are apparently reduced 0.66 to 0.98 percent per meter (0.2 to 0.3 percent per foot) away from the maximum concentration observed at site C-1 in this wetland. It is apparent that much of the iron in the AMD has been deposited in the 3.9 ha (9.6 ac) area surrounding the AMD discharge point.

Isolines of dissolved manganese concentrations in the shallow ground-water system show the highest concentrations are associated with sites CD-1 and D-1 (Figure 6-3). The zone of elevated manganese concentrations is generally confined to the area between the F-2 to C-2 line and Highway 200 or roughly 1 ha

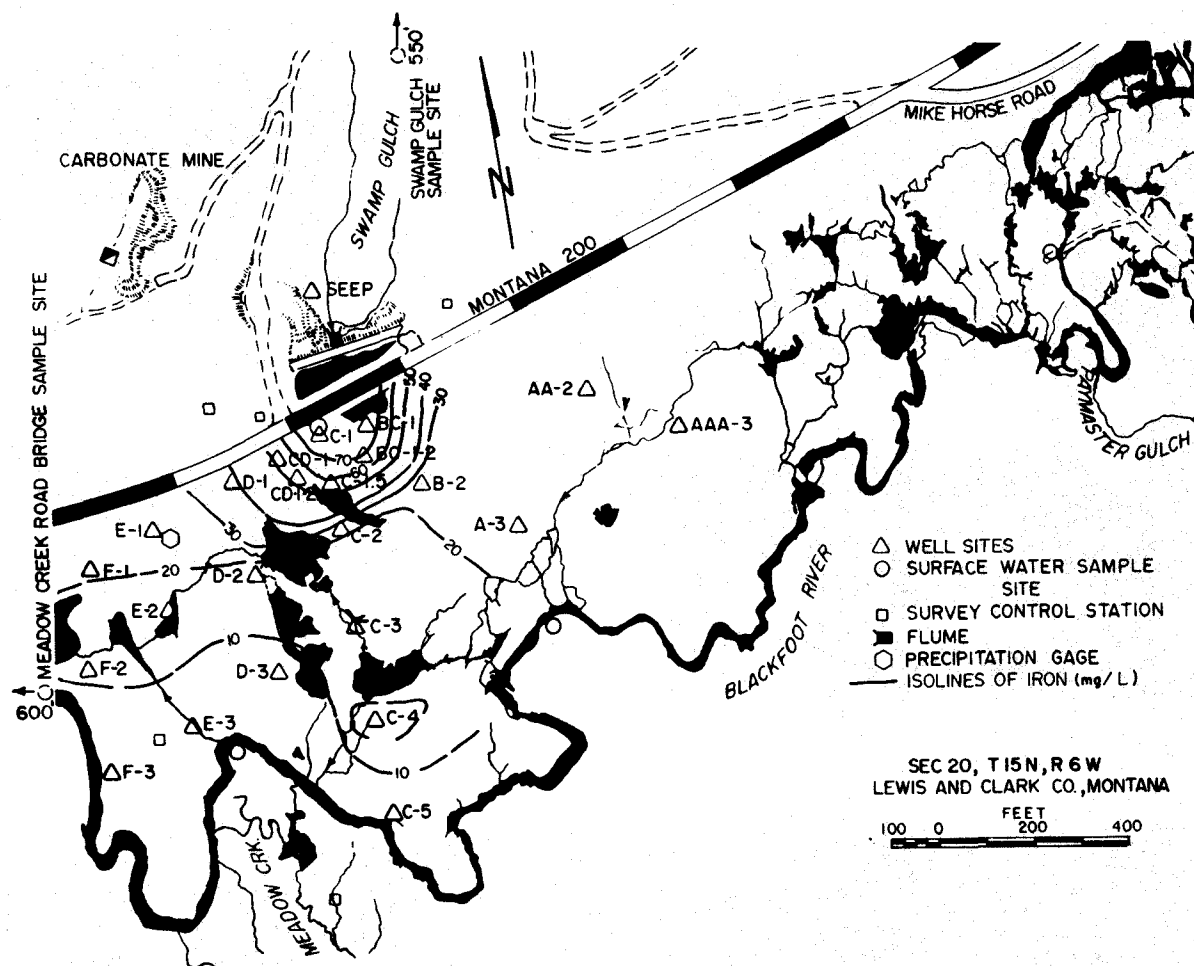


Figure 6-2. Iron concentrations for the wetland shallow ground-water system.

(2.5 ac). Concentrations in this zone are 3 to 11 times background concentrations. High manganese concentrations have also been observed at sites C-5 and E-3 but it is doubtful that these sites are impacted from the Swamp Gulch AMD.

Manganese is apparently attenuated to a much lesser degree than is iron in this wetland. The manganese concentration at site F-1, near the wetland outflow, are still about 4 times above background levels. These results are consistent with other studies which have shown *Sphagnum* wetlands are more effective in removing iron than they are for removing manganese from AMD (Wieder and Lang 1986, Burris et al. 1984).

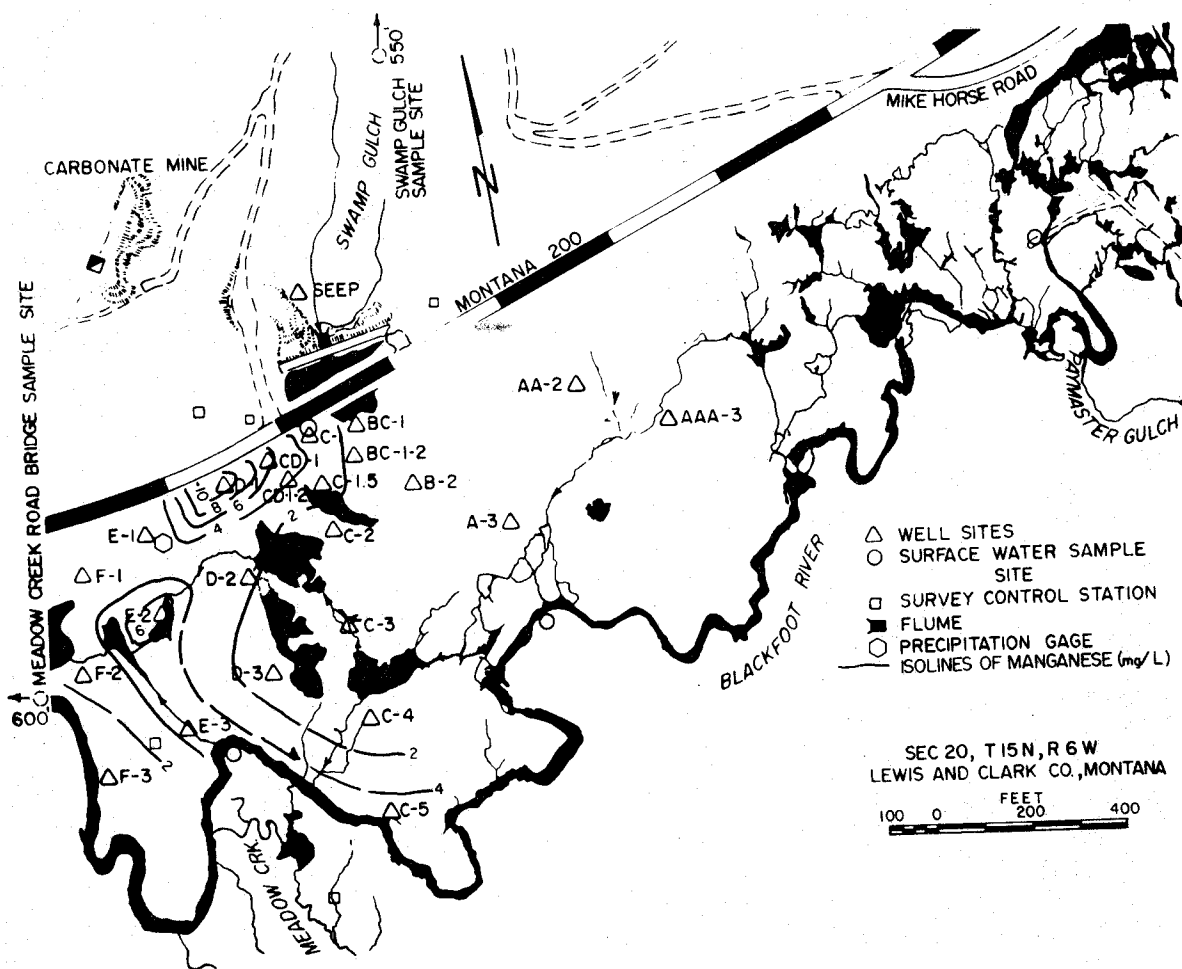


Figure 6-3. Manganese concentrations for the wet and shallow ground-water system.

Isolines of sulfate concentrations follow a pattern very similar to that found for the bromide tracer (Figure 6-4). From a high concentration zone in the D-1 to C-1 area, levels decrease gently towards sites E-1 and B-2, apparently along the preferential flow paths. Sharp concentration reductions over short distances occur at the margins of the flow paths. Concentrations are typically 3 times above background levels in the 0.7 ha (1.8 ac) AMD impacted zone. The sulfate isolines suggest a low level of AMD impact does occur in the area defined by site B-2, A-3, C-3 and C-2. This is supported by the increase in iron and manganese concentrations observed in the

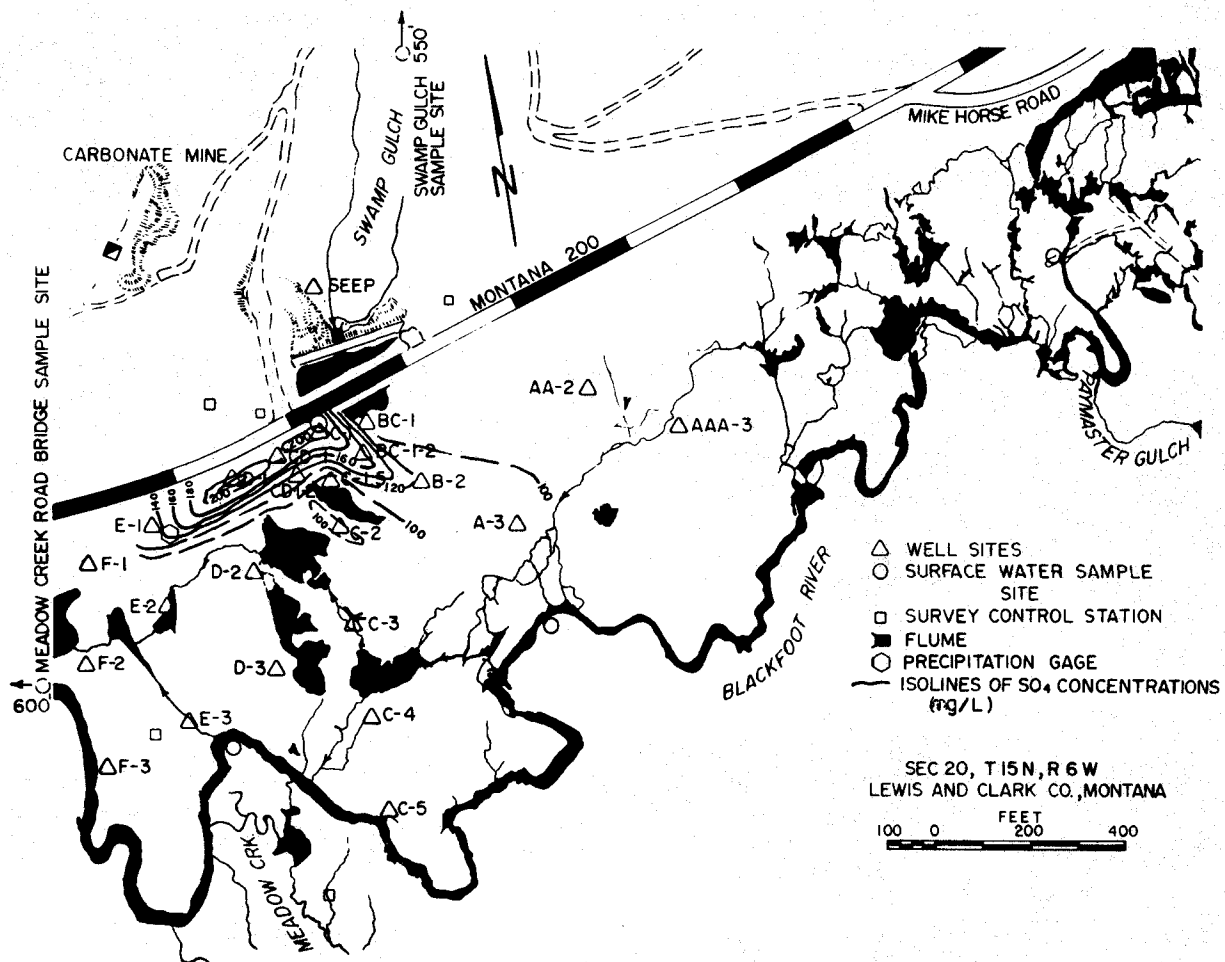


Figure 6-4. Sulfate concentrations for the wetland shallow ground-water system.

surface water sample taken at site C-3 (section 6.4.5). This wetland is apparently effective in attenuating shallow ground-water sulfate concentrations to levels approaching background values.

Considerable variation was apparent in TDS concentrations within the wetland. Mean values ranged from a low of 119 mg/L at site D-2 to a high of 403 mg/L at site D-1. Higher TDS concentrations were generally confined to a 0.6 ha (1.6 ac) area in the same general vicinity where elevated manganese concentrations were found (Figure 6-5). Total dissolved solids concentrations are reduced from values about 2 times above

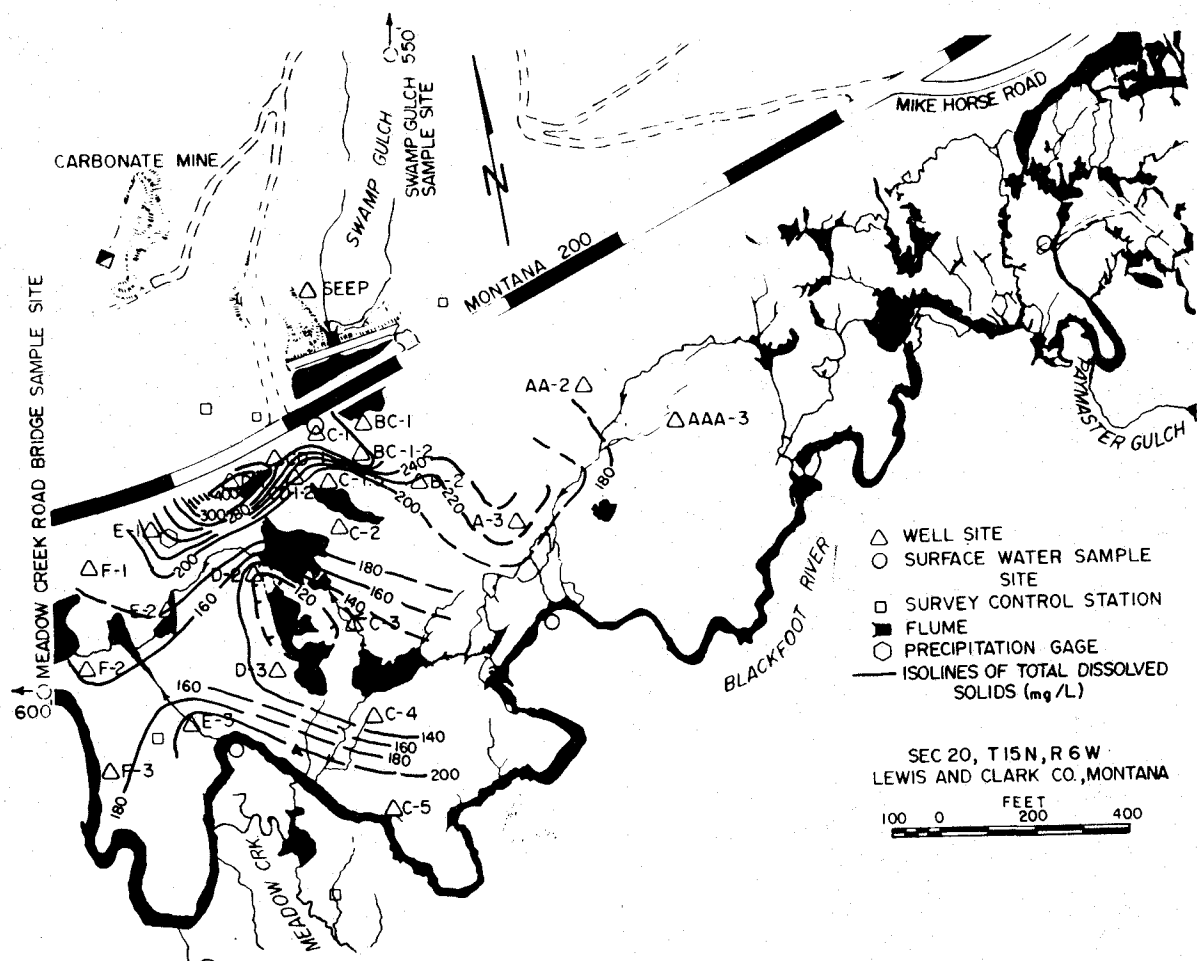


Figure 6-5. Total dissolved solids concentration for the wetland shallow ground-water system

background values to levels similar to background concentrations within the wetland area.

Concentrations of aluminum, cadmium, lead and zinc were generally low throughout the wetland. The maximum aluminum value was 2.8 mg/L in well BC-1-S for April, 1987 sample trip. Aluminum was detected in 18 other samples from 7 other wells which all but one sample exhibited values less than 1 mg/L. Only two samples contained cadmium in excess of 0.01 mg/L, the recommended criteria for potable and irrigation water supplies. The two samples were obtained from sites BC-1-2 (0.032 mg/L) and BC-1-S (0.044 mg/L) both of which are in close proximity to the Swamp Gulch AMD discharge point. These two wells were also the

source of the only two samples in which detectable levels of lead were found (both 0.03 mg/L). All zinc concentrations were well below applicable criteria for all common uses. The maximum zinc concentration in the impacted area was 0.49 mg/L from well D-1-S for the April, 1987 sample trip. The maximum zinc value observed in the wetland was 1.18 for site C-5-S, but this site is not likely affected by the Swamp Gulch AMD.

Numerous other trace metals were analyzed in the initial April 1987 samples. These included arsenic, chromium, cobalt, copper, nickel, antimony, tin, selenium and silver. No shallow wells had values above detection limits for arsenic, chromium, copper, nickel, selenium and silver. Only site C-5-S had detectable cobalt at 0.09 mg/L. Antimony concentrations at or above the 0.01 detection limit were found at 14 sites. The highest antimony levels found were 0.03 mg/L at BC-1-2, D-1-S and C-5-S. These data suggest there is some enrichment of antimony due to the Swamp Gulch AMD, but the effect is minor.

In summary, the levels of aluminum, cadmium, lead, zinc, sulfate and TDS observed in the wetland shallow ground water system should have little overall adverse impact on vegetation or wildlife. However, the high iron and manganese concentrations observed exceed all applicable water use criteria and can be expected to adversely impact all species not adapted to high levels of these metals. The elevated cadmium concentrations observed near the AMD discharge point may also be of concern and may be detrimental to wildlife through bioaccumulation (Section 9).

6.5.2 Deep Ground-water System

The overall water type for the deep ground-water system exhibited markedly less variation than the shallow ground-water system. A calcium-magnesium sulfate water type was dominant in all sites except AAA-3, B-2 and C-2, which were characterized by an iron-calcium sulfate water type, and sites D-1 and F-1 in which calcium-iron bicarbonate and calcium-magnesium bicarbonate dominated, respectively. Site C-3-D exhibited a sodium-iron sulfate water type which was likely the result of problems with bentonite contamination during construction.

Isoline map evaluations for concentrations of iron, manganese, sulfate and TDS suggest the AMD has a much more limited impact on this aquifer system. Only iron and manganese show increases in concentrations associated with AMD and the effect is generally subdued as compared to the shallow aquifer system (Figures 6-6 and 6-7). The areas in which iron and manganese levels are clearly elevated above background concentrations roughly coincide with the impact area observed for elevated TDS levels in the shallow ground-water system and cover approxi-

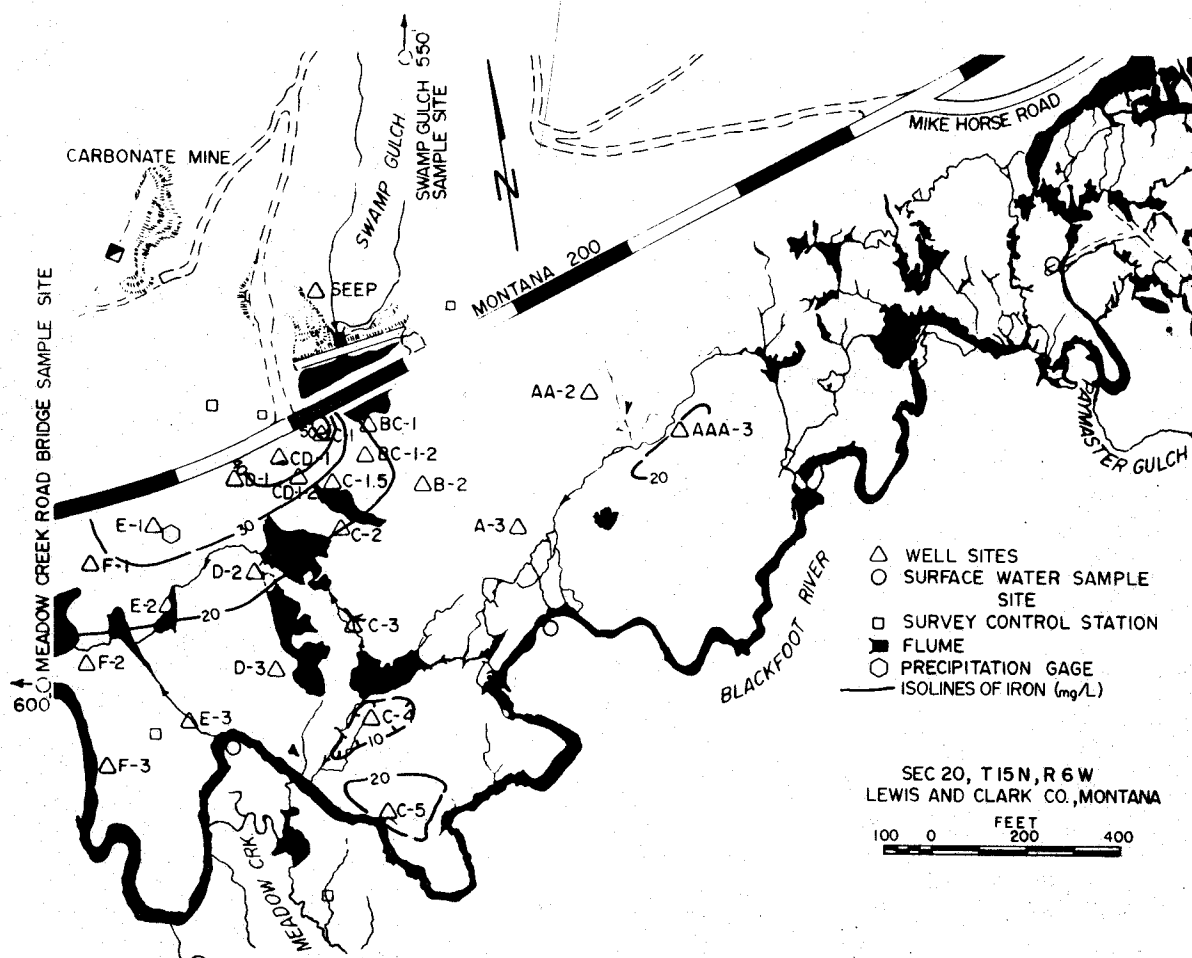


Figure 6-6. Iron concentrations for the wetland deep ground-water system.

mately 1.5 ha (3.6 ac) and 1.7 ha (4.3 ac), respectively. Concentrations and attenuation gradients in the deep ground-water system for both iron and manganese are considerably less than in the shallow ground-water system which suggests a very limited AMD input to the deep system. This conclusion is supported by sulfate and TDS concentrations which were clearly elevated above background levels only at well C-1-D.

No deep wells exhibited lead values above the 0.01 mg/L detection limit. The highest wetland aluminum values from deep wells were found at sites AA-2 and B-2 with maximums of 1.7 and

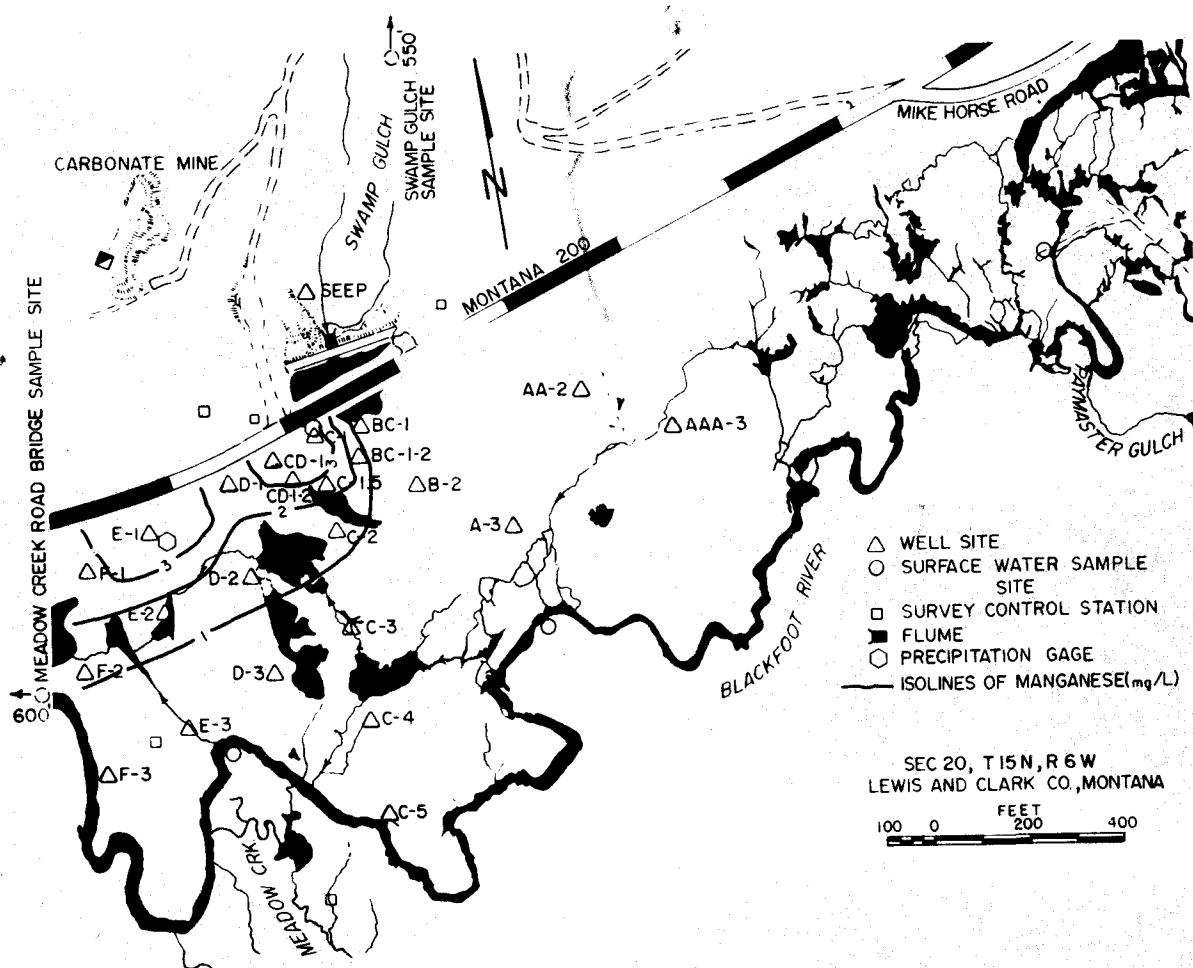


Figure 6-7. Manganese concentrations for the wetland deep ground-water system.

3.0 mg/L, respectively. No other sites were found with aluminum values greater than 1.0 mg/L and many were below the 0.1 mg/L detection limit. Detectable cadmium concentrations were found at sites AAA-3, AA-2, B-2, BC-1, C-1 and C-2. Maximum values were consistently found for site AA-2 with a mean and maximum of 0.007 and 0.012 mg/L, respectively. This was the only site exhibiting levels above 0.01 mg/L, a recommended criteria level for potable and irrigation water. As with shallow ground-water samples, arsenic, chromium, cobalt, copper, nickel, antimony, selenium and silver were determined on the April 1987 sample set. Arsenic was detected at only two sites, AAA-3 at 0.012 mg/L and AA-2 at 0.005 mg/L. Cobalt was detected only at site C-5 which contained 0.02 mg/L. Copper was present in detectable quantities only at sites AA-2 (0.02 mg/L) and B-2 (0.07 mg/L). The maximum antimony concentration found (0.03 mg/L) was for

site C-1. Sites C-5 and D-1 both exhibited antimony values of 0.02 mg/L and all other sites were at or below the 0.01 mg/L detection limit for antimony. No detectable amounts were found for chromium, nickel, selenium and silver in the deep wells. With the exception of antimony, the secondary trace metals have no apparent relationship with the AMD.

It is possible that some of the observed effect of elevated concentrations of iron and manganese in deep wells is due to subsurface seepage from old mine workings that migrates below the highway in fractured bedrock rather than vertical percolation through the wetland muck layer. There is insufficient data at present to evaluate this factor.

6.5.3 Acid Mine Drainage Seepage Wells

The water quality of the shallow and deep "seep" wells is important as it represents the actual water quality that is likely emanating from old mine workings prior to dilution by surface water (Figure 5-9 and 5-10). The general water type for this site is a calcium sulfate with only small to moderate differences noted for most major parameters between deep and shallow samples.

This water was characterized by low pH (3.0) and moderate to high levels of most trace metals. Mean values for TDS, EC and sulfate are 1838 mg/L, 1924 umhos/cm and 1228 mg/L, respectively. Parameters which did exhibit notable differences between shallow and deep samples were aluminum, iron, manganese, zinc and several minor metals. Aluminum and zinc were consistently higher in shallow samples, 70.4 mg/L versus 3.6 mg/L and 31.9 mg/L versus 8.5 mg/L, respectively. This trend was also noted for arsenic, cadmium, copper, nickel and antimony. Both iron and manganese were higher in the deep well than in the shallow well (57.5 versus 5.0 mg/L and 65.1 versus 40 mg/L, respectively). Silver was also apparently present at slightly higher concentrations in the deeper well at this site. It is readily apparent that many of the trace element concentrations present in water at these two wells are quickly diluted by high quality upper Swamp Gulch water before entering the wetland. This process likely eliminates most of the toxicity problems that might otherwise occur. The high levels of many metals, sulfate and fluoride would make this water unsuitable for any common use including potable supplies, livestock, irrigation and aquatic life usage.

6.6 SUMMARY

Of the parameters analyzed (Appendix B), only aluminum, cadmium, copper, iron, lead, manganese, zinc and sulfate were significantly elevated in the Swamp Gulch watershed drainage due

to AMD. Concentrations of these parameters in the wetland shallow ground-water system are elevated to levels above the concentrations found in the AMD near the AMD inflow point and decrease both laterally and vertically from this point. The only catotelm area notably affected by AMD is in the immediate vicinity (30 m - 100 ft) of the AMD discharge point and, in general, this stratigraphic unit is only minimally impacted by the AMD.

Minor changes in the water quality of the Blackfoot River above and below the wetland discharge point suggest that manganese and possibly other several metals may not be effectively ameliorated within the wetland. However, the Swamp Gulch wetland area is at least partially effective in removing iron from the AMD. These data suggest about half of the 21 mg/L weighted mean iron concentration in the mean 45m³ daily AMD inflow may be precipitated within the 3.9 ha (9.6 acre) iron impacted acrotelm area (Section 8). These data suggest some means of physio-chemical precipitation of iron should be incorporated in artificial wetland construction. This system should provide aeration of the AMD to enhance iron precipitation, possibly through the use of waterfalls. Since optimum treatment of AMD pH and sulfate by microbial action (Section 10) requires reduced conditions, the aeration stage should be constructed near the discharge end of the artificial wetland, under the assumption that it will be much easier to add oxygen to the AMD than it would be to remove it.

7.0 DISSOLVED METAL LOADING OF A NATURAL WETLAND

An estimation of the AMD metal load supplied to the Swamp Gulch wetland through the April 1987 - March 1988 year has been made based on water sample concentrations and estimated flow volumes. A comparison of this loading with observed metal masses in the acrotelm (Section 8) permit an estimation of past wetland efficiency in amelioration of AMD.

7.1 METHODS

The calculation of metal loading into the wetland was hampered by the short duration of runoff records necessitating estimations be made for a number of factors. The assumptions that have been made are:

- o the Swamp Gulch input represents all metal loading to the study area;
- o the three sample dates, April 29, July 29, 1987 and January 11, 1988, are representative of all flows for the periods April 1 through June 30, 1987, July 1 through November 16, 1987 and November 17, 1987 through March 31, 1988, respectively;
- o the observed flow on April 29, 1987 is representative of the entire month; and
- o that the mean daily flow rate for the period May 1 through June 15, 1987, was equal to the value for the period June 15 through June 30, 1987.

The latter assumption is based on the Lincoln Ranger Station precipitation which was similar for the two months at 4.1 cm and 3.4 cm (1.62 and 1.33 inches) for May and June, respectively. The Lincoln Ranger Station records also indicate 0.99 cm (0.39 in) of precipitation fell during April, 1987, and that precipitation for December, 1986, January and February 1987 were far below normal. Casual field observations during April and May indicated little snow melt runoff occurred and thus the assumption was made that the April 1987 flow estimate was reasonable for the entire month. The time intervals were determined on the general flow regime and the time of sampling. These are, in general, spring flow consisting of some snow melt and precipitation; summer flow consisting of base flow and precipitation runoff; and, winter low flow (base flow) when no runoff and little precipitation input occur and during which freezing conditions persist.

The wetland metal loading figures are thus based on a limited record for a period in which abnormally low precipitation conditions prevailed. Extrapolation of these data to cover the annual period of April 1987 through March 1988 may involve appreciable error and further extrapolation to other years is not generally recommended.

7.2 RESULTS AND DISCUSSION

The approximate weights (mass) of dissolved metals discharged via Swamp Gulch into the wetland from April 1987 through March 1988 varied from 338 kg (744 lbs) for iron to values less than 100 grams (3.5 oz) for arsenic and silver (Table 7-1). High values for aluminum (107 kg), manganese (63 kg), zinc (44 kg) and copper (12 kg) are also evident. All other metal loading values were less than 1 kg and of these, only cadmium with its ability to be readily bioaccumulated, may be of additional concern. The implications and fate of these metals is discussed further in section 8.

Table 7-1. Total dissolved metal loading (kg) on the Swamp Gulch wetland, April 1987 through March 1988.

	----- PERIOD -----			
	Apr.1-June 30	July 1-Nov.16	Nov.17-March	Annual
Flow Volume(m ³)	7,940	7,360	1,110	
<u>Parameter</u>				
Al	48	52	6.1	107
As	<0.04	<0.04	<0.01	<0.09
Cd	0.302	0.213	0.027	0.542
Cr	<0.159	----	---	----
Co	<0.079	<0.074	<0.011	<0.164
Cu	6.274	4.931	0.410	11.6
Fe	146	138	5.	338
Pb	.278*	0.221	0.044	.543*
Mn	30.8	28.8	3.75	63.4
Ni	0.32	0.29	0.03	0.64
Sb	0.08	<.07	<0.06	0.08
Se	<0.04	----	----	----
Ag	<0.04	<0.04	<.01	<0.09
Zn	22.6	17.9	2.9 cst.	43.6
Annual Total				565 kg

*Estimate

8.0 WETLAND GEOCHEMISTRY AND THE CONTROL OF ACID MINE DRAINAGE

The Swamp Gulch wetland area core samples have been analyzed for numerous chemical and physical parameters (Appendix C). These analyses were made to determine 1) the extent of past metal loading; 2) the distribution of various parameters with depth and stratigraphic changes; and 3) the basis for comparison of elemental concentrations observed in vegetation growing on this site.

8.1 METHODS

Samples for geochemical evaluation were obtained during construction of well and auger hole installations. Core samples were obtained by methods outlined in section 5.2.1. These samples were placed in plastic sample bags and shipped to Energy Laboratories in Billings, Montana. Samples for metal analyses were digested using hydrogen peroxide, EPA Method 3050 (EPA 1982). The very high iron content of most wetland soil samples necessitated special corrections to eliminate interference in samples analyzed by inductively coupled argon plasma spectroscopy (ICP). These corrections were calculated at the 25 percent iron content and, as a result, were insufficient for the very high iron content found at site C-1. The apparent concentrations reported for several metals are depressed at site C-1 for this reason but the laboratory reported values are still included in Appendix C. Another result of the iron interference was the high tin and antimony values (positive interference) reported in the initial screening samples upon which the selection of parameters to be analyzed was based. As a consequence, tin and antimony were run on samples from several sites and, following interference corrections, all values were below the instrument detection limit. Two National Bureau of Standards (NBS 1982) samples of river sediment, standard 1645, were run with wetland samples. Recoveries were within 10 percent for all parameters except aluminum, arsenic and iron. The recoveries were: aluminum, 28%; arsenic, 79%; cadmium, 90%; chromium, 97%; copper, 92%; iron, 79%; mercury, 100%; manganese, 92%; nickel, 90%; lead, 95%; and zinc 94%. The low recoveries for aluminum make interpretation of these data difficult and the findings should, therefore, be considered only relative. No adjustment of these data has been made since, with the exception of aluminum, most elements were within or very near the reported range of estimated uncertainties for the standard. Chloride, fluoride, bicarbonate and sulfate were analyzed from a saturated paste extract. The cation exchange capacity was determined by Method 8-4, American Society of Agronomy Monograph 9 (1965) for acid soils. Organic carbon content was calculated from loss on ignition at 550°C for 2 hours. The data base consists of single samples for each selected interval at each site, generally based

on field observed stratigraphic changes. Fourteen laboratory duplicates and the two standards were analyzed for most parameters.

Selected metal deposition in the wetland due to the Swamp Gulch discharge has been estimated utilizing 1) isopachs of the acrotelm (Figure 5-7), 2) isoline maps of chemical concentrations of specific metals, and 3) summation of volumes multiplied times concentrations multiplied times bulk density. Background concentrations were subtracted before isoline maps were constructed for this purpose.

8.2 RESULTS AND DISCUSSION

In general, metal and other chemical parameters are more varied than similar data for ground-water which is likely due to a more heterogeneous nature of the soils. Occasional anomalously high or low values for several elements have been noted and may be due to sampling or laboratory ICP interference. Samples obviously in error have not been used in the following interpretation. Total masses of metals deposited in the acrotelm due to AMD have been calculated based on the 3.9 ha (9.6 ac) area found to be impacted by iron deposition. The total volume of the impacted acrotelm area is approximately 23,000 cubic meters (30,000 yd³).

Most trace metal concentrations in the acrotelm show similar areal distribution. Aluminum, cadmium, copper, lead, manganese and zinc all exhibit the highest concentrations in the area immediately south of Montana Highway 200 and generally west of site C-1. Iron distribution more closely follows the preferential ground-water flow paths (Section 5). The following paragraphs describe the distributions of specific metals in the wetland soil materials.

8.2.1 Areal Distribution of Selected Elements

8.2.1.1 Iron

Iron concentrations in the wetland acrotelm vary from a maximum of 415,000 ppm at site C-1 to 36,400 ppm at site AA-2 (Figure 8-1). The distribution pattern is similar to the ground-water flow paths found during the tracer study which strongly suggests the Swamp Gulch drainage is the source of most of the iron found in the acrotelm. Concentrations of iron fall from the maximum (about 8 times background levels) to values about 2 times above background levels at site F-1. This indicates the wetland has been effective in past removal of iron from the AMD.

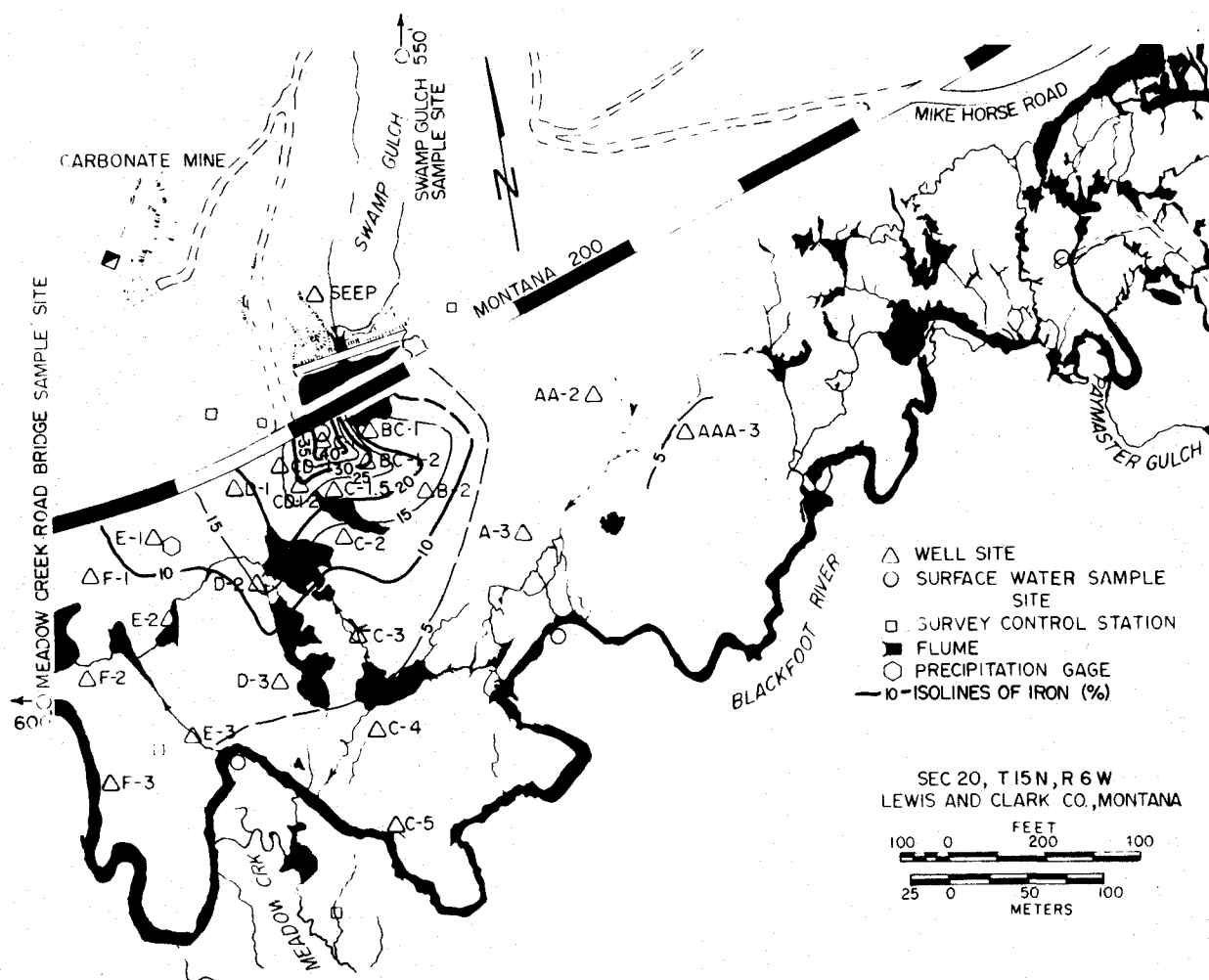


Figure 8-1. Iron concentrations in the wetland acrotelm.

An estimation of acrotelm iron mass due to the Swamp Gulch AMD discharge indicated approximately 550 metric tons (600 U.S. tons) of iron were deposited from this source. This amount suggests the present annual estimated iron loading [(340 kg/year (750 lbs/year - Section 7) has been greatly exceeded in the past. It is likely that iron discharges to the wetland were notably higher during the active mining/milling period and that the annual estimated amount may have been underestimated due to abnormal climatic conditions during the study period.

8.2.1.2 Manganese

The distribution of manganese in shallow wetland soils is marked by high concentrations in the site D-1 to site F-1 area (Figure 8-2). Values at site D-1 are approximately an order of magnitude above background sites AA-2 and AAA-3. Concentrations of soil manganese decrease with increasing distance from site D-1 suggesting some manganese is being removed from the groundwater and sequestered in the acrotelm. However, the manganese concentration at site F-1 is still three times greater than background levels which indicates a notable amount of manganese may likely be discharged from the wetland. Approximately one metric ton (2,400 lbs) of manganese has been deposited in the wetland acrotelm from the Swamp Gulch AMD. The boundary for the elevated manganese impacted area was less well defined than that

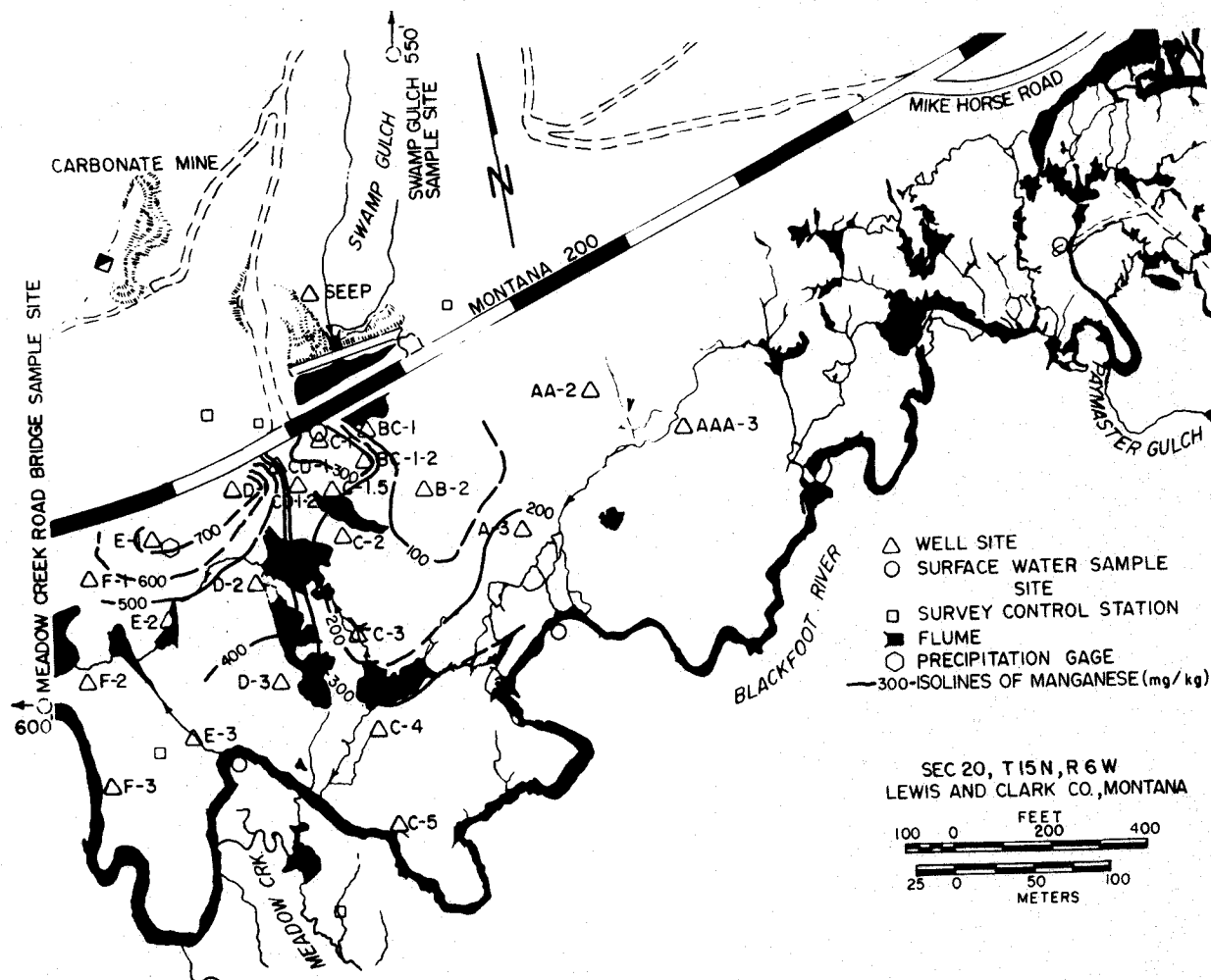


Figure 8-2. Manganese concentrations in the wetland acrotelm.

for iron and, therefore, the one ton estimate may be conservative. However, it is clear that the efficiency of this wetland in the removal of manganese is at least an order of magnitude less than for iron.

8.2.1.3 Aluminum

The relationship between the Swamp Gulch discharge and elevated aluminum levels in the acrotelm are the least distinct of the metals evaluated, possibly due to ICP iron interference as suggested by the low recoveries found in the NBS standards. Relative concentrations at several sites (BC-1, C-2, D-1, E-1, E-2 and F-1) are apparently two to three times higher than background levels at AAA-3, AA-2 and A-3 (Figure 8-3). However, the reported aluminum values at sites CD-1, CD-1-2, and C-1.5, all close to the Swamp Gulch discharge point, were similar to

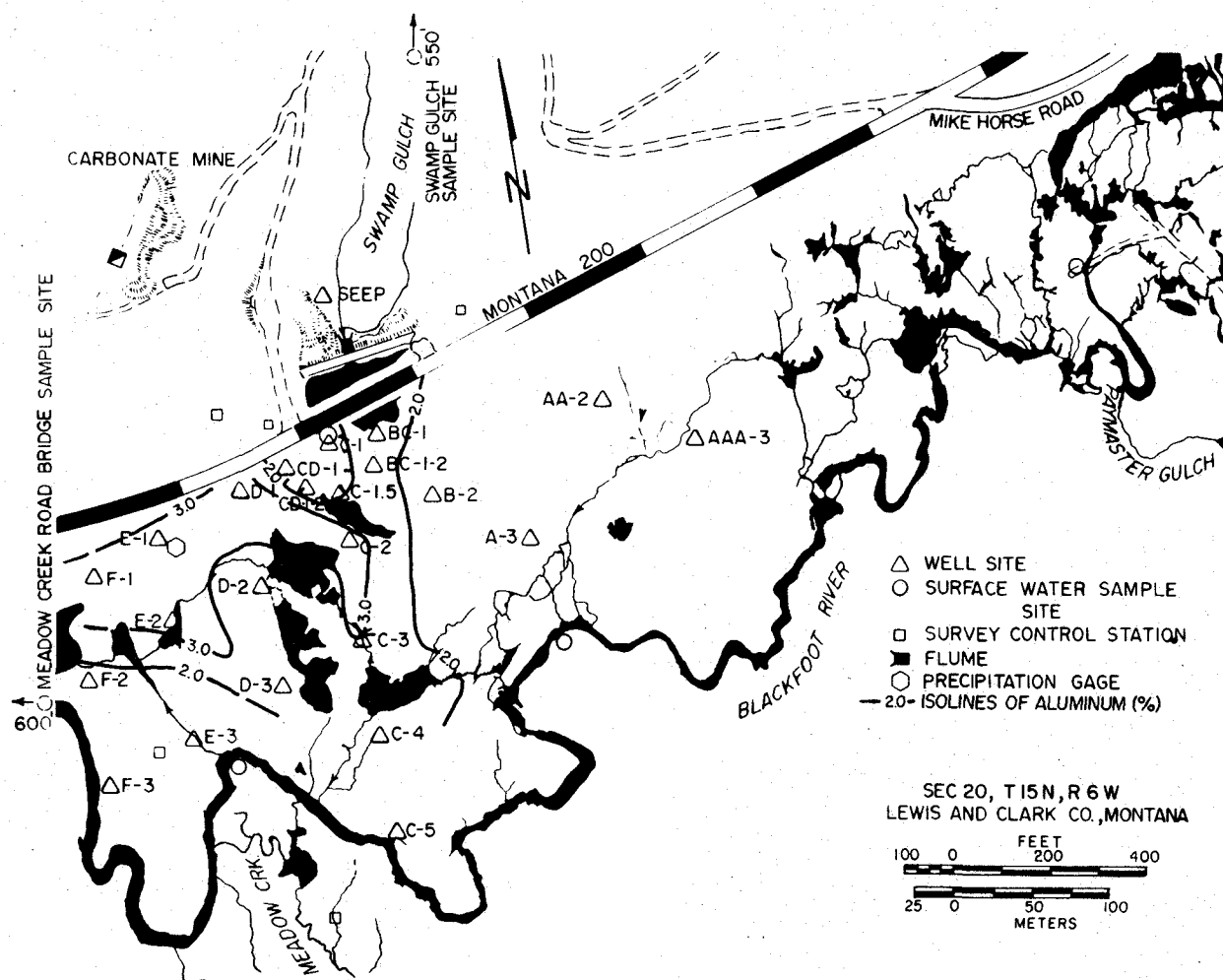


Figure 8-3. Aluminum concentrations in the wetland acrotelm.

background concentrations. The relative aluminum levels in the acrotelm are consistently higher to the west of the C-1 to C-5 line than they are upgradient to the east. It is possible that iron interference may have produced the low aluminum values reported at sites CD-1, CD-1-2 and C-1.5 and may have produced lesser uncorrected interference with all samples. Due to the apparent problems evident with aluminum analyses, no estimation of the aluminum mass present in the acrotelm has been attempted.

8.2.1.4 Copper

Copper levels in the wetland acrotelm exhibit a marked increase in concentration near the Swamp Gulch discharge point. Background concentrations as represented by sites AAA-3, AA-2 and A-3 range from 210 mg/kg to 44 mg/kg. Sites B-2 and B-3 also exhibit low copper concentrations at 19 and 40 mg/kg, respectively. Acrotelm copper values rise abruptly west of a line running due south from site BC-1, which roughly corresponds to the 100 mg/kg isoline (Figure 8-4). The highest concentrations are found in the C-1.5 to F-1 zone where copper levels are generally 20 to 30 times the background mean. The remaining wetland area to the south of this zone is characterized by values 6 to 8 times the mean background concentration. The distribution pattern suggests some amelioration of copper has occurred in this system, but the high levels still evident at site F-1 suggest much copper may pass through the system and discharge to the Blackfoot River. The acrotelm copper mass is about four metric tons (4.4 U.S. tons) which represents 325 years of deposition at the 1987-88 annual rate of 11.6 kg (25.5 lbs). While this period would be well in excess of the period from mine initiation to the present, past loading rates were likely higher (as observed with iron and lead deposition) and hence, removal of copper from the AMD is likely considerably less than 100 percent. The 40 to 50 mg/kg range, typical of the wetland background areas, is common in numerous agricultural soils (EPA 1987b). Other studies have found soil total copper levels of 500 to 700 ppm to produce severe yield reductions in rye grass and bush beans (Wallace et al. 1977, McGrath et al. 1982). Numerous other crop species are significantly impacted at soil total copper levels as low as 100 ppm (EPA 1987b).

8.2.1.5 Lead

Lead concentrations in the wetland acrotelm vary from less than 1 mg/kg to values greater than 1000 mg/kg. Background values upgradient from the Swamp Gulch discharge point are generally less than 100 mg/kg but are higher than the 11.6 to 32.2 mg/kg range that Chattopadhyay and Jervis (1974) found for the Holland Marsh muck soil. The anomalously high level (450 mg/kg) found at background site AAA-3 is likely the result of other influences and not the result of the Swamp Gulch AMD discharge. The wetland system is apparently quite efficient in

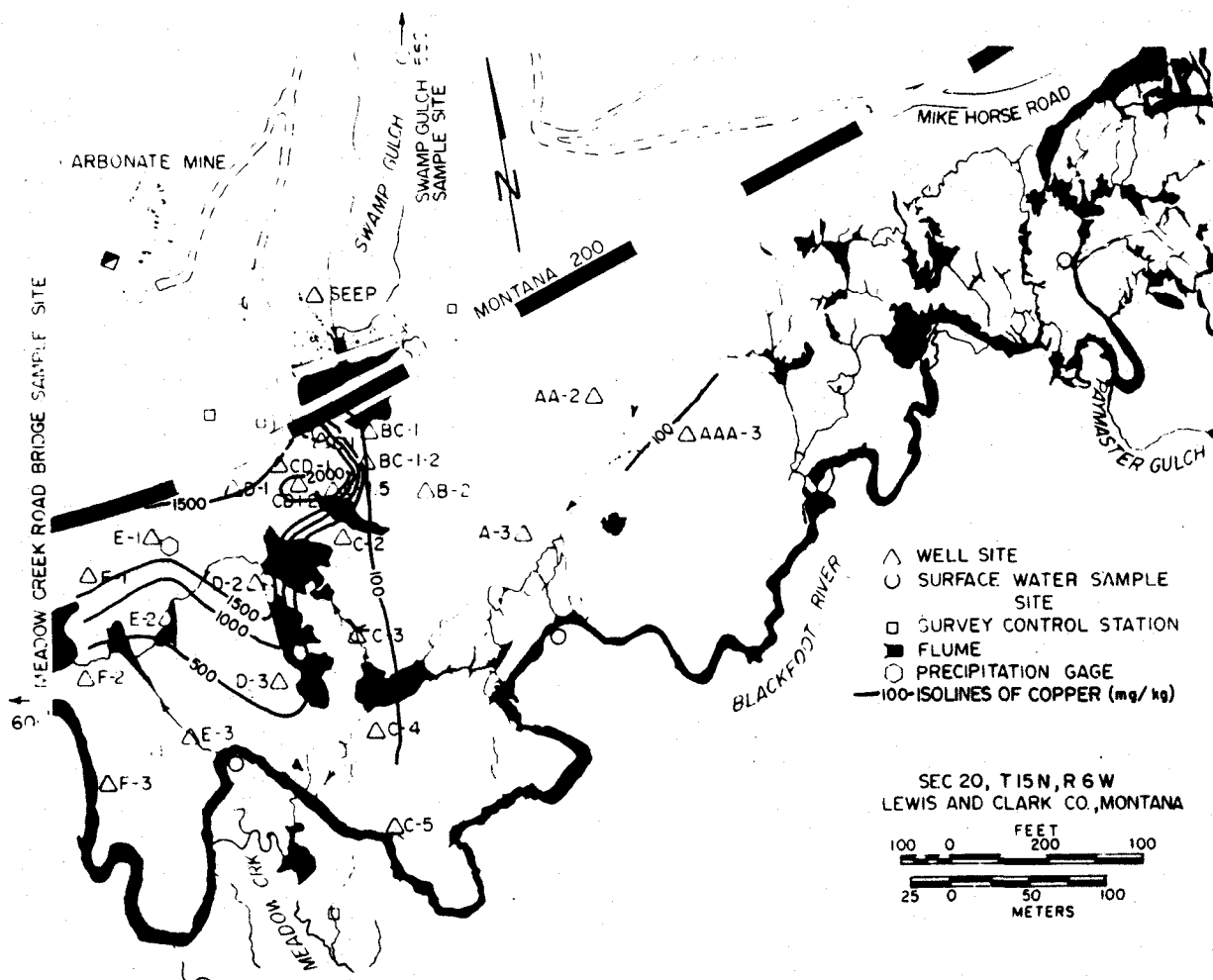


Figure 8-4. Copper concentrations in the wetland acrotelm.

the removal of lead, likely due in part to the low mobility of lead in soils. The Swamp Gulch acrotelm contains a AMD derived lead mass of 1.3 metric tons (1.4 U.S. tons) which represents 2,350 years of deposition at the calculated 0.54 kg (1.2 lbs) annual loading rate. It is clear that past lead loading rates were higher than those observed during this study. While some of the observed lead may have been derived from natural mineralization in the area, it is doubtful that much lead from this source would have been deposited in the acrotelm due to the semi-isolation of the acrotelm provided by the more limited

hydraulic conductivity of the underlying catotelm and the limited mobility of lead. Elevated levels found in the zone roughly described by sites C-1, C-1.5, D-3 and E-1 are an order of magnitude above background sites. The concentrations at sites E-2 and F-1 are approaching background levels. Of particular interest are the high lead concentrations found in near surface materials at sites F-2 and F-3 (Figure 8-5). These are likely the result of mining activities on the upper Blackfoot River above this site. A literature review on the phytotoxicity of selected metals (EPA 1987a) suggests lead concentrations observed in the elevated zone would be phytotoxic to most crops and may inhibit maximum vegetation production in the wetland.

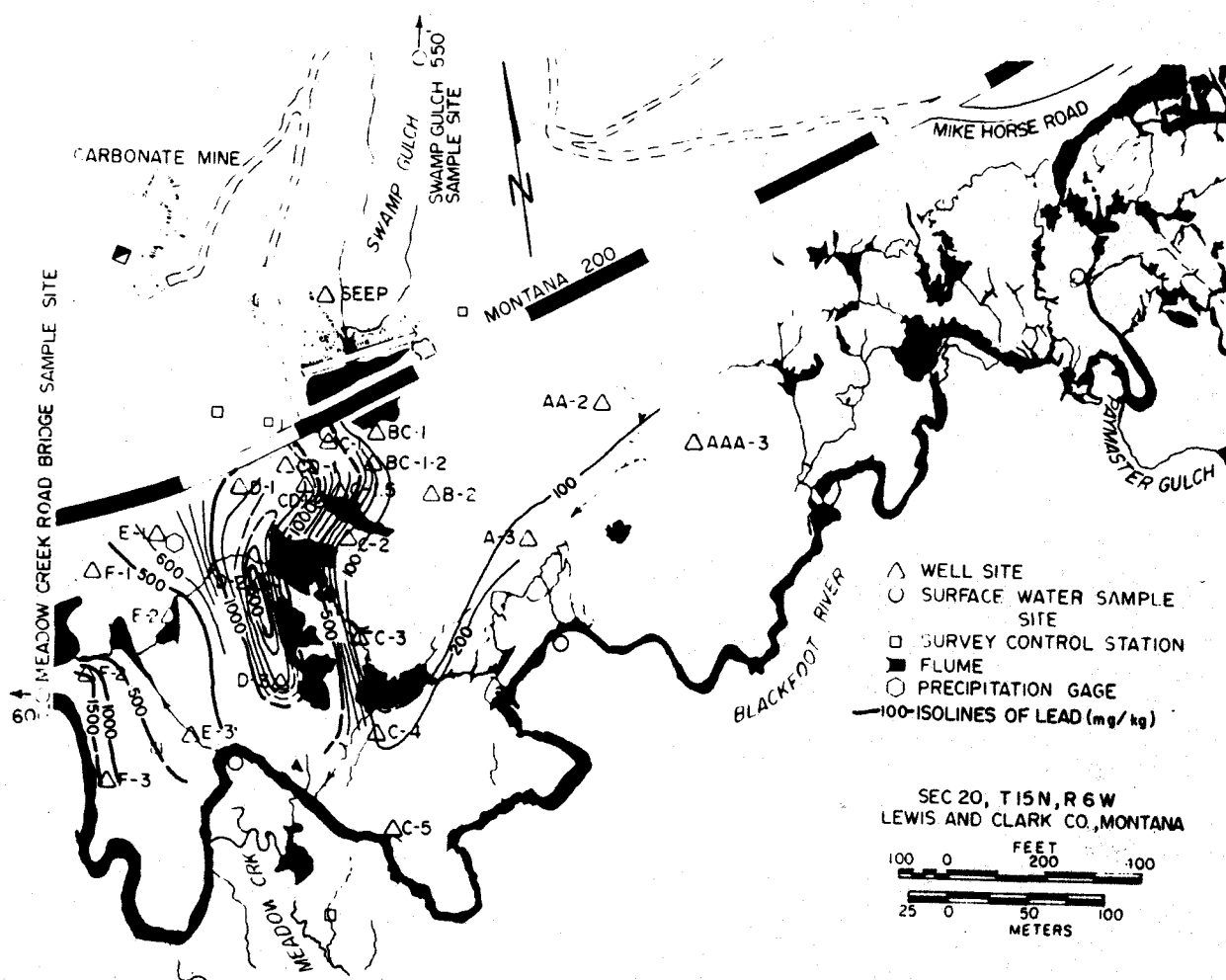


Figure 8-5. Lead concentrations in the wetland acrotelm.

8.2.1.6 Zinc

Zinc concentrations in the wetland acrotelm exhibit several anomalies, including high levels at background sites AAA-3 (1780 mg/kg) and A-3 (2092 mg/kg) and a low concentration at site C-1. The cause of these variations is not apparent but may be due in part to analytical interference. It is evident that zinc concentrations in the acrotelm are consistently elevated south of Montana Highway 200 from sites C-1.5 to F-1 (Figure 8-6). Zinc concentrations in this area are generally an order of magnitude higher than most background levels. It is also evident that zinc concentrations are not very effectively reduced down-gradient from the Swamp Gulch discharge point.

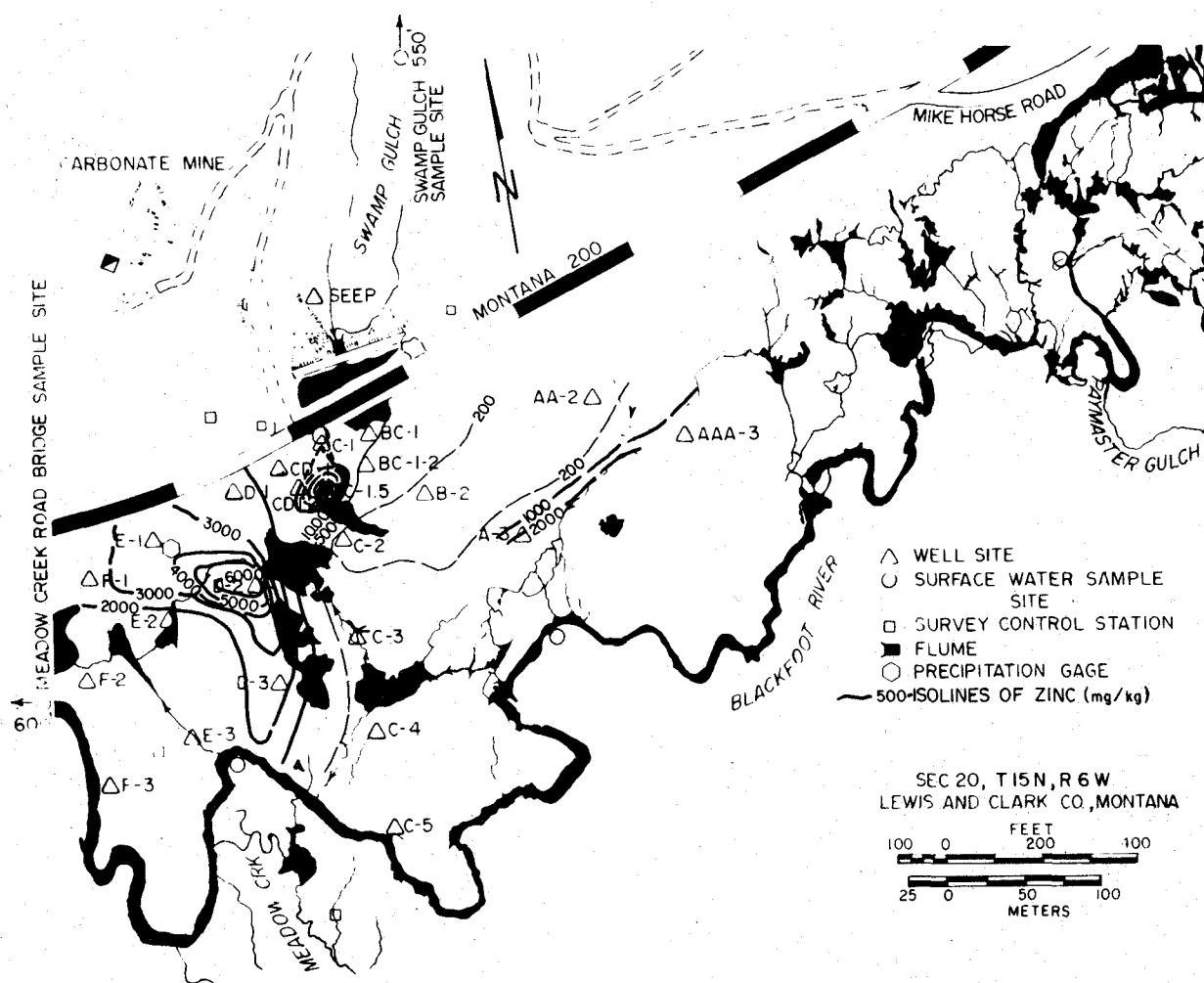


Figure 8-6. Zinc concentrations in the wetland acrotelm.

Concentrations of zinc in the elevated zone are consistently above 1500 mg/kg, a level that would be expected to result in nearly a 100 percent yield reduction for most agricultural crops (EPA 1987a).

The Swamp Gulch wetland acrotelm zinc mass due to AMD is approximately 6.0 metric tons (6.6 U.S. tons) which would represent 137 years of deposition at the observed 43.6 (95.9 lbs) annual loading rate. Although this time interval is in excess of historic mining and AMD discharges, past discharges of zinc were undoubtedly higher and hence, the actual time interval represented is likely considerably less.

8.2.1.7 Arsenic

Arsenic concentrations at the four wetland sites analyzed for this element (AA-2, C-1, C-4 and F-1) exhibit similar distribution patterns with depth. All sites are characterized by highest concentrations located at the top of the muck zone. The maximum arsenic value is 86 mg/kg for the 0.46 to 0.91 m interval at site C-4. Values at AA-2 and C-1 are similar and suggest the wetland acrotelm has little effect on AMD arsenic content at this study site. Total arsenic levels as low as 10 ppm have been shown to be detrimental to production for some crop species (EPA 1987a). Surficial acrotelm values range from 18.5 mg/kg at site AA-2 to 29.5 mg/kg at site C-4, both of which sites are not likely impacted by the Swamp Gulch AMD. Some impact to wetland vegetation and wildlife may occur from the observed arsenic concentrations.

8.2.1.8 Barium and Beryllium

Barium and beryllium distributions are generally similar to arsenic. These two elements are more generally distributed within the muck zone with the highest concentrations tending more towards the central area of this stratigraphic zone. A notable exception to this observation occurred at site C-1 where the maximum beryllium concentration (240 mg/kg) occurred in the acrotelm, strongly suggesting this element is enriched due to the Swamp Gulch discharge. The maximum barium concentration (2140 mg/kg) was found in the 1.77 to 1.89 m depth interval at site F-1.

8.2.1.9 Cadmium

Cadmium concentrations in the wetland soils range from <0.5 to 45 mg/kg. This metal is apparently enriched 10 to 20 times in wetland soils due to the Swamp Gulch AMD. The highest concentrations are associated with the acrotelm layer at site F-1 (44 and 45 mg/kg). At site F-1, the muck zone maximum is 4 mg/kg. This is in contrast to site C-1 where there is apparently little difference between most muck values and the

acrotelm (1.2 to 21 mg/kg and 22 mg/kg, respectively). The AMD derived acrotelm cadmium content is only 4 kg (8.8 lbs) which represents 6.7 years of deposition at present loading rates. These data suggest some Swamp Gulch discharge is percolating through the muck zone at site C-1 and that the wetland is generally less effective in ameliorating cadmium in AMD. At background site AA-2, the cadmium concentration in the acrotelm is 2 mg/kg and a maximum concentration of 8.6 mg/kg is associated with the gravels beneath the muck zone. All muck zone values at this site are less than 2 mg/kg. The 22 to 45 mg/kg values found in the acrotelm at sites C-1 and F-1 can be expected to severely impact production of many crops (EPA 1987a) and can be expected to add significant cadmium to the food chain, possibly impacting wetland wildlife (Section 9).

8.2.1.10 Cation exchange capacity

Cation exchange capacity (CEC) of the acrotelm determined on sites AA-2, C-4 and F-1 (Appendix C), had a weighted mean of 35.6 meq/100g. The range of CEC in the wetland, including site C-1, was 4.9 to 45.0 meq/100g. These CEC findings are lower than those typically found in wetlands. Mathur and Farnham (1985) found wetland peat materials with a mean CEC of 148.5 meq/100g. These low CEC findings in the Swamp Gulch wetland may be the effect of the method of CEC analysis. Presuming the CEC found is true, the low CEC may be a reason for the limited level of wetland effectiveness in ameliorating the AMD metals.

All samples from sites AA-2, C-1, C-4 and F-1 that have been analyzed for mercury, antimony, and thallium are below the 1 mg/kg instrument detection limit for these elements.

8.2.2 Distribution of Elements with Depth

An evaluation of selected trace metals distributions with depth indicates that, in general, the anaerobic muck zone below the acrotelm contains lower concentrations of most elements (Table 8-1).

Aluminum is a notable exception, and is often found at higher concentrations in the muck zone. A comparison between mean background levels and concentrations within the AMD impacted area suggests the muck zone is impacted by AMD (Table 8-1). All metals evaluated in the impacted muck zone were higher than corresponding background values. However, due to the lower hydraulic conductivity of the muck zone, it likely has limited effective remediation potential for AMD amelioration. A comparison of muck zone concentrations with the underlying gravel materials reveals a less clear situation in which, compared to the catotelm, some metals are higher and some lower in the gravels. Zinc levels were often higher in the gravels,

Table 8-1. Mean selected elemental concentration (mg/kg) changes with depth for background and AMD impacted areas.

Site	Al	Cu	Fe	Mn	Pb	Zn
<u>Zone - Acrotelm</u>						
Background (AA-2,A-3)	9,930	46	64,100	225	125	1380
Impacted Zone (C-1,D-1,F-1)	27,150	1287	236,000	898	689	2580
<u>Zone - Catotelm</u>						
Background (AA-2,A-3)	22,300	92	42,300	123	117	188
Impacted Zone (C-1,D-1,F-1)	37,800	597	63,800	468	291	704
<u>Zone - Gravels</u>						
Background (AA-2,A-3)	23,200	285	26,250	95	77	1190
Impacted Zone (C-1,D-1,F-1)	17,830	563	32,870	413	170	693

especially near the top of this unit (Appendix C). Relative sulfate levels, as determined from the saturated paste extracts, suggest the distribution of sulfate is similar to zinc. The highest sulfate values for background sites AAA-3, AA-2 and A-3 are found in the underlying gravels. This distribution is also evident at sites D-1 and D-2. Sites B-2, C-5 and D-3 have the highest sulfate concentrations in the acrotelm. The only site where high sulfate values are associated with the catotelm is C-1. Low sulfate values in the catotelm would be expected due to reducing conditions which would tend to reduce sulfate to insoluble sulfides which would not be recovered in a saturated paste extract. It is possible that the total sulfur concentration in this zone would exceed that in other stratigraphic units. This observation of sulfate removal is substantiated by the studies on microbiological sulfate removal presented in Section 10, Phase II.

8.3 SUMMARY

Annual deposition of metals into the acrotelm have probably decreased from levels present during the period of active mining and milling. Acrotelm masses of lead, iron, copper and zinc would, at present depositional rates, represent accumulation periods ranging from 2,350 to 137 years (Table 8-2). If the assumption is made that the apparent depositional interval (2,350 years) found for lead represents 100 percent removal of this metal, and that this interval can serve as a relative basis of comparison with other elements, the wetland efficiency in removal of metals is: iron, 70%; copper, 14%; zinc, 5.8%; manganese, 0.7%; and cadmium, 0.3% (table 8.2). Assuming past loading rates were higher than those observed at present, it is apparent that cadmium, manganese and likely zinc are all passed through the wetland without much amelioration. Copper may also be passed through the system, but it would appear that it is at least partially ameliorated. Both lead and iron may be effectively removed within the wetland area. Much of the apparent decrease in wetland water sample concentrations of cadmium, manganese, zinc and possibly copper with increasing distance from the AMD discharge point, is probably due to dilution processes rather than immobilization. The higher levels of metals observed in the gravels may be derived from two sources: 1) from the original gravel materials which contain pebbles and grains of mineralized materials derived from upstream outcrops; and 2) from mineralized ground-water flowing laterally in these materials. It is also possible, especially at site D-1, that there could be an upward ground-water gradient beneath the gravels in bedrock materials. This ground-water could be derived in part from seepage emanating from old underground mine workings.

Table 8-2. Acrotelm metal masses and the relative efficiency of wetland metal removal.

	Acrotelm metal mass (metric tons)	Annual Input from AMD (kg)	Years represented by deposition rate	Relative efficiency compared to lead(%)
Cadmium	0.004	0.54	6.7	0.3
Copper	3.8	11.6	325.	14.
Iron	553.	338.	1,640.	70.
Lead	1.3	0.54	2,350.	100.
Manganese	1.1	63.4	16.9	0.7
Zinc	6.0	43.6	137.	5.8

It is clear that most observed sequestering of AMD metals occurs in the acrotelm, with the catotelm playing a minor role. Efforts should be concentrated on constructing effective acrotelm zones in artificial wetlands, and over time, a natural catotelm may form from material derived from the acrotelm.

9.0 WETLAND VEGETATION AND THE CONTROL OF ACID MINE DRAINAGE

The objectives of this portion of the Swamp Gulch wetland study were to:

- 1) quantitatively describe plant associations and determine production by dominant species, and
- 2) assess metal loading of dominant vegetation and potential impacts to wildlife in the natural wetland.

9.1 SAMPLE COLLECTION, PREPARATION AND ANALYSIS

Vegetation observations were recorded during three trips to the Swamp Gulch wetland from April through August, 1987. Plant reference specimens and vegetation samples were collected from the Swamp Gulch study area and from the Hardscrabble Creek background wetland during August 1987. Specimens have been deposited in the Reclamation Research Herbarium.

The composition of the vegetation at the Swamp Gulch site was determined by estimating plant canopy coverage. Eight transects were located across the study area, along which quadrats (20 X 50 cm) were placed at regular, preselected intervals. A total of 104 quadrats were used. Species specific canopy coverage was estimated at each sampling station to the nearest percent (Sensu Daubenmire 1959). Depth to the water table was also estimated at each sampling station. At the background site on Hardscrabble Creek, canopy coverage values for the dominant species were visually estimated for the site as a whole.

Aboveground production for the dominant herbaceous species (Carex rostrata) was estimated at the Swamp Gulch and the Hardscrabble background site using 0.25 m² (0.5 x 0.5 m) quadrats. Plants were clipped at ground level from seven quadrats at the Swamp Gulch site and in two quadrats at the background site. These samples were placed in paper bags, oven-dried at 50° C to a constant weight and then weighed to the nearest 0.1 gram. Values were converted to kg/ha and comparisons were made between the Swamp Gulch site and the Hardscrabble site at the 0.05 level using the Students t-test.

Vegetation samples from throughout the Swamp Gulch area and background sites were collected for chemical analysis. To expedite this process, the Swamp Gulch study area was divided into seven collection zones. From each zone, whole plants were collected for Carex rostrata and the mosses, Isopterygium pulchellum and Sphagnum tenellum. Current growing season shoots and leaves were collected from dominant shrub species (Salix

boothii and Betula glandulosa), as were roots up to approximately 3 cm in diameter. Aboveground materials were placed in paper bags and below ground materials were placed in plastic bags. Initial sample preparation consisted of washing each sample with tap and distilled water and then subjecting it to ultrasound until no residues were observed in the water. Each sample was then oven-dried (50° C), placed in paper bags and, together with National Bureau of Standards (NBS) quality control samples, were shipped to Energy Laboratories in Billings, Montana on September 28, 1987. A total of 67 samples (65 natural and 3 NBS citrus leaf samples) were analyzed for Al, As, Cd, Ca, Cu, Fe, Pb, Mn, Ni and Zn. At the laboratory, each sample was digested with nitric acid and hydrogen peroxide according to method 3050 of the EPA Manual SW-846 (EPA 1982).

Concurrent with vegetation sampling, a map was drawn delimiting the plant associations, areas of open water, transect locations, and man-made features such as roads.

Canopy coverage and frequency data were summarized for the Swamp Gulch study area as a whole. Nomenclature follows Dorn (1984) for vascular plants and Crum et al. (1973) for mosses. The final plant association map was constructed with the aid of the field map, field notes, canopy cover data, and after a review of wetland classification schemes made by other authors.

9.2 DATA QUALITY ASSURANCE/QUALITY CONTROL

The quality of vegetation chemical data were determined for both field and laboratory procedures. Field sampling included the collection of duplicate samples and the insertion of NBS Blind Field Standard (BFS) reference samples into the sample train. Laboratory QA/QC procedures consisted of splits of natural and field duplicate samples. Analytical results of these samples were used to calculate precision and accuracy statements following standard procedures developed for the EPA Contract Laboratory Program. Appendix D-1 presents the results of the data accuracy and precision statements.

Accuracy is interpreted through the following example: for Al, we are 90% confident that the reported value is $44.5 \pm 6.3\%$ of the true value. Precision is interpreted as follows: for Al, we are 90% confident that the true value is within $\pm 4.5\%$ of the reported values.

9.3. RESULTS AND DISCUSSION

9.3.1. Vegetation Patterns

The Swamp Gulch wetland was a mosaic of plant associations, the presence of which was directly related to flooding frequency and duration. Land surface varied from areas of continually open water (about 4.6% of the site) to areas where the soil was periodically saturated (sometimes nearly year round), to areas where the water table was rarely high enough to saturate the entire soil profile. The wettest areas were characterized by pure stands of Carex rostrata (Figure 9-1). This species was nearly ubiquitously present, being associated with Betula glandulosa and Salix boothii, and with Pinus contorta. Carex rostrata was found growing as an emergent (in as much as 20 cm of water) and where the soil surface was only damp. A relatively dry site was occupied by Pinus contorta, Chimaphila umbellata and Sphagnum tenellum. The driest areas were dominated by species indicative of the surrounding forest; these areas were characterized by Pinus contorta, Picea englemannii, and Arctostaphylos uva-ursi. A list of vascular plants and bryophytes identified within the Swamp Gulch wetland study area can be found in Appendix D-2. Beaver dams, extending across the study area near transect 6 (Figure 9-1), contributed to the patterning of plant associations. The eastern two-thirds of the site had a higher water table than the western one-third, and was therefore dominated by Carex rostrata and associated communities. The western portion was dominated by Pinus contorta and forest grasses and forbs.

9.3.2. Canopy Coverage and Production

Carex rostrata was the dominant species throughout the study area. It was found at over 90 percent of the sampling stations and covered more than 55 percent of the study site (Table 9-1). Betula glandulosa was the dominant shrub and Pinus contorta was the dominant tree. Salix boothii was also important; it covered 5.5 percent of the study area and was found at almost 20 percent of the sampling stations. Two mosses, Sphagnum tenellum and Isopterygium pulchellum were commonly encountered, especially in the wetter, eastern two-thirds of the site. Chimaphila umbellata was associated almost exclusively with Pinus contorta in a small area adjacent to Highway 200 (Figure 9-1). Sphagnum tenellum grew profusely within this association.

Many dead Pinus contorta can be observed, especially in the eastern portion of the wetland. These were from 20-30 feet tall and probably from 40-60+ years old. The most likely cause of death was a rise in the water table, which may have been the result of fluctuating climatic conditions.

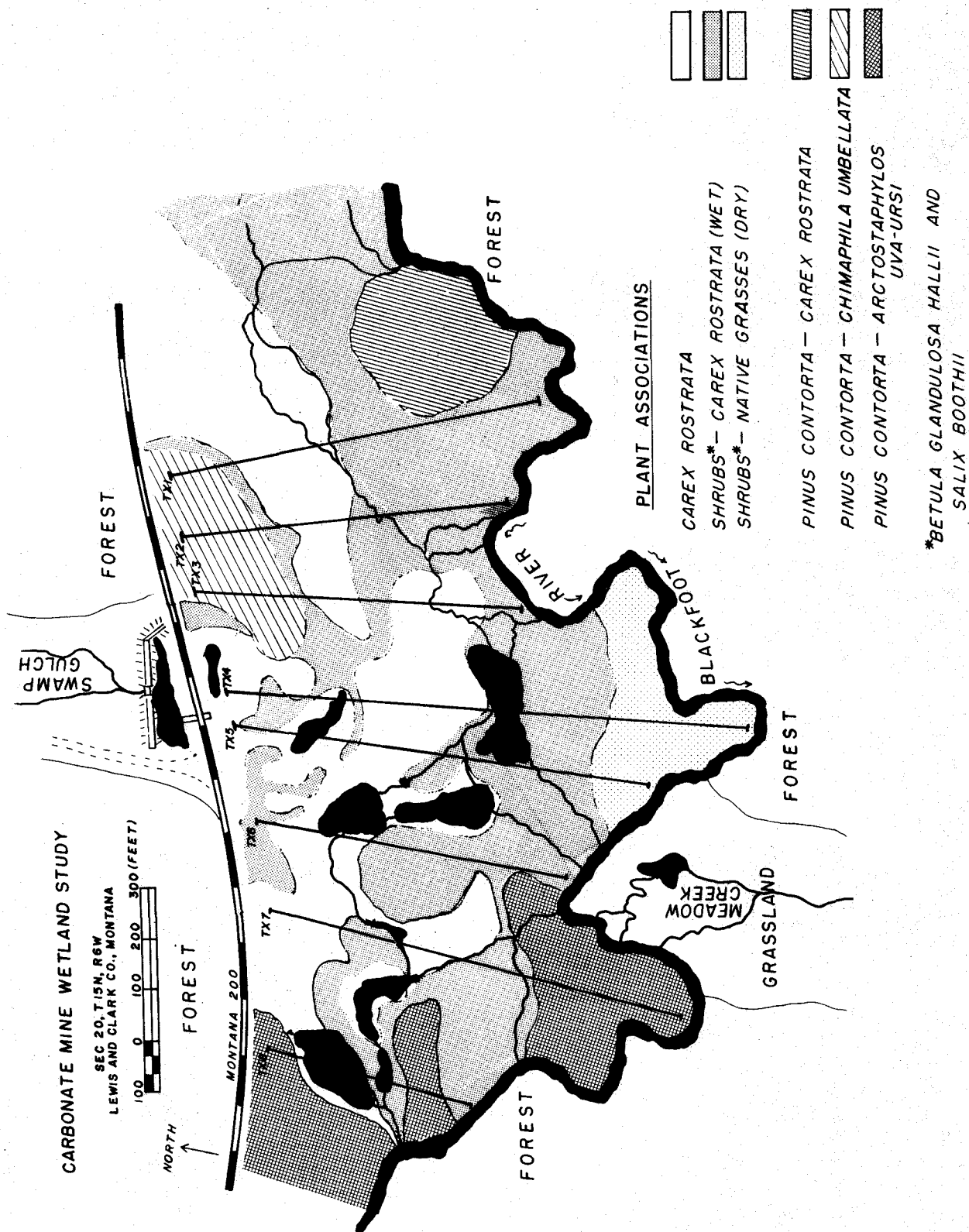


Figure 9-1. Vegetation associations and canopy coverage transects.

Table 9-1. Mean percent canopy coverage and percent frequency for plant species.

Species	Life Form ¹	% Canopy Coverage	% Frequency
Carex rostrata	GL	55.5	90.4
Betula glandulosa var. hallii	S	11.3	31.7
Pinus contorta	T	6.3	8.7
Salix boothii	S	5.5	19.2
Sphagnum tenellum	M	4.7	12.4
Chimaphila umbellata	S	3.0	8.7
Potamogeton sp.	F	2.8	1.9
Isopterygium pulchellum	M	2.8	18.1
Pyrola asarifolia	F	1.2	7.6
Arctostaphylos uva-ursi	HS	1.1	5.7
Abies lasiocarpa	T	0.9	1.0
Juniperus communis	S	0.9	1.9
Picea engelmannii	T	0.9	4.8
Scirpus acutus	GL	0.9	1.2
Pseudotsuga menziesii	T	0.6	1.0
Equisetum sp.	F	0.5	11.5
Vaccinium scoparium	HS	0.3	1.0
Achillea millefolium	F	0.1	1.9
Elymus glaucus	G	<0.1 ²	1.0
Fragelia virginiana	F	<0.1	4.8
Festuca idahoensis	G	<0.1	1.0
Geum macrophyllum	F	<0.1	1.0
Phleum pratense	G	<0.1	1.0
Unknowns		2.2	1.0
TOTAL		101.8	

¹ LF = Life Form: GL = Grasslike, S = Shrub, T = Tree, M = Moss, F = Forb, HS = Half-Shrub and G = Grass

² Values of <0.1 were tabulated as 0.05.

Herbaceous production on the Swamp Gulch wetland averaged 3820 kg/ha (Table 9-2). For comparison, vegetation composition and production were estimated at a pristine background wetland on Hardscrabble Creek (off the Alice Creek road). The composition of the vegetation was very similar to the Swamp Gulch wetland. Carex rostrata dominated with about 60 percent canopy coverage and Salix boothii and Betula glandulosa were the dominant shrubs. Water temperature, pH and EC were 11^o C, 7.4 and 400 umhos/cm, respectively. Herbaceous production at the background site was 3750 hg/ha, which was not significantly

Table 9-2. Peak standing crop (kg/ha) at the Swamp Gulch and Hardscrabble Creek (background) wetlands.

Hardscrabble Creek (background)			Swamp Gulch Wetland		
Mean ¹	STD	Range of Values	Mean	STD	Range of Values
3750a	±532	3374 - 4127	3820a	±1550	1983 - 6776

¹ Means were not significantly different at the $p \leq .05$ level.

different than production on the Swamp Gulch site (Table 9-2).

9.3.3 Element Concentrations in Plant Material

This section presents data on the concentration of elements in the dominant plant species at the Swamp Gulch study site and at the Hardscrabble Creek (background) wetlands. These data are compared, contrasted and discussed with respect to water quality data.

Statistical analysis was conducted to determine if elemental differences in plant tissues could be detected between the Swamp Gulch and the background sites. Large variations in the analytical data (and low sample numbers) resulted in large standard deviations. This situation restricted the usefulness of the statistical analyses by limiting the number of possible significant determinations. As a consequence, element data are discussed in a general manner without the use of statistics.

A literature search revealed no data on the metal levels of the plant species occupying the study site. Considerable data exists however, for species of cattails (*Typha*). Hutchinson (1975) reviewed the literature on metal levels in aquatic vegetation. EPA (1987a and b) conducted a literature review on metal concentrations in terrestrial macrophytes for the purpose of establishing plant and animal hazard criteria. These data sets will be compared, in a general way, with element data in the tissues of the macrophytes at the Swamp Gulch and Hardscrabble Creek wetlands. Metal levels in a variety of bryophytes, particularly *Sphagnum* spp., growing in metal-enriched areas have been reported. These data will also be compared with the bryophytes metal data from the Swamp Gulch and Hardscrabble sites.

Water quality data are presented for three locations at the Swamp Gulch site. This is done to show the degree of metal enrichment in water as it flows past the mine site, and to help explain the level of metal attenuation by the wetland.

Background water chemistry levels were established from samples collected in Swamp Gulch above the Carbonate Mine. References will also be made to the water quality at station F-1, located directly down-gradient from the Swamp Gulch wetland. These data, therefore, should provide information as to the quantity of metals being attenuated by the wetland.

Table 9-3. Concentration (ug/g) of elements in Carex rostrata.

Element	Matrix ¹	Background Wetland		Swamp Gulch Wetland	
		Mean(n=4)	Range	Mean(n=5)	Range
Al	AG	61	11-170	232	41-400
	BG	408	7 ² -1085	1720	850-2670
As	AG	0.9	0.35 ² -2.1	0.9	0.6-1.4
	BG	16	1.8-35	19	1.5-41
Cd	AG	0.8	0.35-1.6	0.9	0.5-3.5
	BG	6.2	0.6-21	14	6.5-17
Ca	AG	3285	360-4530	2456	1110-3450
	BG	6420	2340-15000	3500	670-5150
Cu	AG	3.8	0.35 ² -8.7	62	0.35-160
	BG	30	5.1-78	984	88-2240
Fe	AG	1327	350-3840	6134	2440-15300
	BG	20800	5070-58500	37140	24800-54000
Pb	AG	6.5	1.0-22	29	5.2-70
	BG	128	2.0-490	354	190-800
Mn	AG	248	81-460	250	160-430
	BG	177	88-380	131	43-190
Ni	AG	0.7 ²	0.7-0.7	0.8	0.7-1.2
	BG	1.9	0.7-5.5	9.3	0.7-17.0
Zn	AG	96	14-320	134	69-290
	BG	1096	42-4210	1296	310-2700

¹Plant material: AG=above-ground, BG=below-ground

²Values below the instrument detection limit were estimated by multiplying the instrument detection limit value by 0.7.

9.3.3.1 Effects of *Carex rostrata* in Remediating AMD

Below-ground tissues had higher concentrations than above-ground tissues (Table 9-3). This was true for the background and the Swamp Gulch wetlands and for all the elements with the exception of Mn, which appeared to be more concentrated in the leaves than in the roots/rhizomes. These findings were consistent with previous research on metal uptake by species of *Typha* (Bayly and O'Neill 1972, Heil and Kerins 1988, Taylor and

Crowder 1983a, 1983b). In the present study, above-ground tissues concentration were different from below-ground concentrations by an order of magnitude for Al, As, Cd, Cu, Fe, Pb and Zn.

Water quality samples collected down-gradient from the Swamp Gulch wetland indicated that some of the metals were being attenuated by *Carex rostrata*. Water at site F-1 had lower concentrations of Al, Cd, Cu, Fe, Pb, Ni and Zn than the influent water (Table 9-4). Sulfate was also lower and pH was unchanged.

Some elements (Al, Cu, Fe and Pb) were substantially elevated in plant tissues from the Swamp Gulch site, compared to background levels. These elevated levels were apparently not

Table 9-4. Water quality¹ in Swamp Gulch above and below the Carbonate mine site, and below the Swamp Gulch wetland at station F-1.

Parameter	Above Carbonate Mine (background)	Below Carbonate Mine (AMD)	Swamp Gulch Wetland (Station F-1)
Al	<0.1	6.2	0.4
As	<0.005	<0.005	<0.005
Ca	12.7	22.0	26.0
Cd	0.001	0.038	<0.001
Cu	<0.01	0.61	0.04
Fe	0.7	25.8	15.1
Pb	<0.01	0.035	<0.01
Mn	0.08	3.73	4.30
Ni	<0.03	0.37	<0.03
Zn	<0.01	2.64	0.12
SO ₄	10.7	202.3	142.0
pH	6.8	3.6	3.3

¹Values are means from three sampling dates: 4/29/87, 7/28/87 and 1/11/88.

phytotoxic. Carex rostrata did not appear to be under abnormal stress at the Swamp Gulch site; production (3820 kg/ha) and cover (56 percent) were high, and individual plants appeared to be healthy. As previously discussed, production was statistically similar between the two sites and no difference in canopy cover was visually apparent. Phytotoxicity for Carex rostrata is discussed by element in the remainder of this section.

Although Al was elevated in plant tissue (approximately four times background) at the Swamp Gulch site, the above-ground mean of 232 ug/g fell within the range for aquatic plants reported by Hutchinson 1975 (Table 9-5). The below-ground tissue concentration of 1720 ug/g, however, was substantially higher than data from non-enriched areas reported by other researchers (Table 9-5). Heil and Kerins (1988) reported similar results for Typha latifolia in a wetland constructed in Montana to control AMD; tissue concentrations of Al were higher from the area receiving AMD than from a control site. The uptake of Al by Carex rostrata may have contributed to the low quantity (0.4 mg/L) of Al measured in water flowing from the Swamp Gulch wetland (Table 9-4) at station F-1. The most reasonable explanation for the higher levels of tissue Al in the Swamp Gulch wetland was that Carex rostrata has the ability to absorb this element if it is available. Water quality data showed that Al was substantially higher in the AMD entering the

Table 9-5. Range and mean element concentrations (ug/g) in aquatic forbs and grasses (from Hutchinson 1975).¹

<u>Metal</u>	<u>Range</u>	<u>Mean</u>
Al	250-785	366
As	2.8-20 ²	--
Cd	2.6-28	8.0
Cu	2.5-243	48
Fe	70-31,500 ³	3170
Pb	2.0-53	11
Mn	100-23,000	2380
Ni	1.1-44	--
Zn	26.5-1000	143

¹Except where noted, these values are from non-enriched environments. However, some of the plant species investigated may be metallophytes.

²From As-enriched environment.

³This range may be biased high due to analyses of vegetation that were not completely cleaned of Fe precipitates.

wetland compared to water above the Carbonate Mine site (Table 9-4). This indicates, at least to some degree, that Al will be absorbed by Carex rostrata.

The average concentration of Fe in the influent AMD (25.8 mg/L) was 37 times higher than the water above the Carbonate Mine site (0.7 mg/L). Although these data are limited, the concentration of Fe in water at station F-1 was 15.1 mg/L, which was less than the concentration in the influent water (25.8 mg/L). Tissue concentrations for Fe in the Swamp Gulch wetland were 4.6 (above-ground) and 1.8 (below-ground) times higher than the background. This suggests that a portion of the dissolved Fe was being absorbed by Carex rostrata. Iron precipitates adhering to organic matter also contribute to a reduction in dissolved Fe content as the water passes through the wetland. Leaf tissue levels of Fe at the Swamp Gulch wetland were within the range reported for aquatic plants (Table 9-5). Bayly and O'Neill (1972) reported that Typha latifolia had leaf tissue Fe levels from 200 to 1800 ug/g and rhizome Fe levels from 700 to 3600 ug/g, depending on the season in which samples were collected. In a metal smelting area, Fe concentrations in Typha latifolia leaves and roots reached 905 and 57,138 ug/g, respectively (Taylor and Crowder 1983b). These data were comparable to those measured in the Swamp Gulch wetland, as were values reported by Heil and Kerins (1988). These researchers reported Fe concentrations in leaf tissue of 286 ug/g and root tissue levels of 40,884 for Typha latifolia growing in an artificial wetland receiving AMD.

Like Al and Fe, the concentration of Zn in water increased as it flowed past the Carbonate Mine site (Table 9-4) and was likewise attenuated as the water passed through the wetland. The influent concentration of Zn was 2.64 mg/L, while the concentration at station F-1 was only 0.12 mg/L. This large decrease was not due to absorption by Carex rostrata since tissue concentrations were only slightly greater at the Swamp Gulch wetland compared to the background wetland (Table 9-3). Discussion in the following sections will show that the shrubs accumulated substantial amounts of Zn. Tissue levels in Carex rostrata from both sites were similar to aquatic plant data from non-enriched areas (Table 9-5). In a metal-enriched environment, root and leaf Zn concentrations in Typha latifolia ranged from 13 to 101 ug/g and from 24 to 572 ug/g, respectively (Taylor and Crowder 1983b). These researchers also reported that tissue levels of Zn were not correlated with soil/sediment levels. Phytotoxicity of Zn in terrestrial plants varies from 60 to more 800 ug/g depending on the plant species, the variety and the part of the plant analyzed (EPA 1987a). These authors suggested a 50 ug/g tolerable Zn level and a 500 ug/g phytotoxic level for above-ground tissue.

Dissolved Cu increased in Swamp Gulch water as it flowed past the Carbonate Mine site. Copper concentrations in water increased from <0.01 mg/L above the mine to 0.61 ug/L below the mine. Apparently, Cu was attenuated in the wetland since the concentration in water at station F-1 was 0.04 ug/L. Vegetation data indicated substantially higher (16 and 33 times greater than background in above-ground and below-ground tissues, respectively) Cu concentrations in Carex rostrata tissue from the Swamp Gulch wetland compared to the background site, suggesting that this species was absorbing dissolved Cu from AMD. Leaf tissue concentrations in the Swamp Gulch wetland (0.35 -160 ug/g) were within the range reported for aquatic plants (Table 9-5). However, the range of Cu values in the below-ground material (88-2240 ug/g) often exceeded the range for other aquatic plants. From a natural but metal-enriched wetland, Typha latifolia leaf concentrations reached 24 ug/g and root levels reached 265 ug/g (Taylor and Crowder 1983a). The global diagnostic background level for Cu in above-ground tissue of terrestrial plants ranges up to 20 ug/g (EPA 1987b). Phytotoxicity in terrestrial plants generally begins at levels >20 ug/g. Some Carex rostrata plants in the Swamp Gulch wetland had leaf tissue levels up to 160 ug/g, suggesting that some phytotoxicity may be occurring due to Cu. However, Carex rostrata may have an internal exclusionary mechanism that allows it to tolerate relatively high levels of Cu in its tissues.

The concentration of Pb in the wetland influent AMD was 0.035 mg/L, which was slightly greater than <0.010 mg/L in the water at station F-1 (Table 9-4). These data suggest that some Pb (albeit, very little) was being attenuated within the wetland, a portion of which may have been absorbed by the sedge. Carex rostrata Pb tissue levels were 4.5 (above-ground tissue) and 2.8 (below-ground tissue) times higher in samples from the Swamp Gulch wetland compared to the background wetland (Table 9-3). Above-ground tissue concentrations were within the range for aquatic plants reported by Hutchinson (1975) (Table 9-5). It has been difficult to establish phytotoxic levels for Pb in plants because of the great variability between species. Background levels in lettuce leaves, for instance, have been reported as high as 50 ug/g, while phytotoxic levels in alfalfa have ranged down to 10.8 ug/g (EPA 1987a). These authors felt that a leaf Pb level of 25 ug/g could, in general, be tolerated by grain crops in Montana. Since leaf tissue levels ranged from 5.2 to 70 ug/g in the Swamp Gulch wetland, some plants may be under stress due to high levels of Pb. However, it is also possible that Carex rostrata has a mechanism which allows it to tolerate higher Pb levels than crop species.

The lower concentrations of the aforementioned metals (Al, Cu, Fe, Pb and Zn) in water at station F-1 compared to the influent AMD indicates that a portion of these elements were being attenuated within the wetland. Vegetation data suggests

that Carex rostrata absorbed a portion of these metals. Manganese, by contrast, was not accumulated by Carex rostrata even though the concentration of Mn was notably elevated (47 times) in the influent AMD. Concentration in the influent AMD was 3.73 mg/L compared to the background water level of 0.08 mg/L. Both above- and below-ground tissue concentrations were similar between the background and Swamp Gulch wetlands. Apparently, Carex rostrata has an external exclusionary mechanism that prevents the absorption of Mn. Not only was Mn not being absorbed by Carex rostrata, it was not being attenuated in any other manner. The concentration of Mn in water down-gradient from the wetland (at station F-1) was 4.30 mg/L, which was slightly greater than the influent concentration of 3.73 mg/L. In contrast, Typha latifolia can absorb substantial amounts of Mn from water, Heil and Kerins (1988). These investigators reported mean leaf tissue levels of 1053 and 1760 ug/g of Mn in Typha latifolia from two artificial wetlands receiving AMD having an average of 1.35 mg/L of Mn in the influent water. Manganese tissue levels have ranged up to 23,000 ug/g in aquatic plants (Table 9-5). However, these data were from a study conducted in the Ukraine and it is not clear from Hutchinson's discussion whether the samples were from a metal-enriched area or from a metallophytic species.

The concentration of As in the tissues of Carex rostrata were similar between the background and the Swamp Gulch wetlands (Table 9-3). The concentration of As in the Swamp Gulch water did not increase as it flowed past the Carbonate Mine site (Table 9-4). The tissue concentrations observed in the Swamp Gulch wetland were within the range reported for aquatic plants (Table 9-5). Depending on the plant species and the particular tissue analyzed, phytotoxic symptoms in terrestrial plants have been reported from 5 to 20 ug/g As (EPA 1987a). Most of these data were from crop species, which may be more sensitive to As than Carex rostrata. Based on the currently available aquatic and terrestrial plant data, as well as data from the Hardscrabble Creek (background site), there is no reason to suspect that As is toxic to Carex rostrata at the levels observed.

The concentration of Cd in water above the Carbonate Mine was 0.001 ug/g compared to 0.038 ug/g below the mine site (Table 9-4). Mean leaf tissue concentrations were similar between sites but the range of values suggests that Carex rostrata was absorbing and accumulating more Cd at the Swamp Gulch wetland than the background wetland. Leaf tissue levels at the background site ranged from 0.35 to 1.60 ug/g, compared to 0.5 to 3.5 ug/g at the Swamp Gulch wetland. Tissue levels from this study were similar to those of other aquatic plants (Table 9-5). A large body of data exists on Cd toxicity in crops. Background levels for Cd have been reported up to 3.1 ug/g in alfalfa tops and yield increases have been reported when Cd was present at much higher concentrations (EPA 1987a). In a study of hazardous

levels in crops from a metal enriched area, 10 ug/g Cd in plants was considered tolerable while a tissue level of 50 ug/g would definitely cause a yield reduction. In light of these hazard levels and the data from the Swamp Gulch site, the levels of Cd in Carex rostrata appear to be normal and well below the phytotoxic level. It should be noted that while these levels may not be phytotoxic, they are at least an order of magnitude higher than concentrations found in the AMD water and represent notable bioaccumulation.

Like most of the other elements, Ca was more concentrated in the roots/rhizomes than in the leaves (Table 9-3). As previously mentioned, Ca levels were higher in tissues from the background wetland compared to the Swamp Gulch wetland. This was most likely due to a greater availability of Ca at the background site, although the water and soil at that site were not analyzed. Tissue Ca levels from the Swamp Gulch wetland were within the ranges reported by Taylor and Crowder (1983b) for Typha. Bayly and O'Neill (1972) reported similar levels for rhizomes (2000-9000 ug/g) but higher values for leaves (5000-11,000 ug/g), compared to Swamp Gulch wetland values.

The concentration of Ni in leaf and below ground material in the Swamp Gulch wetland were probably not elevated above background (Table 9-3). These levels fell within Hutchinson's (1975) range for aquatic plants (Table 9-5). Taylor and Crowder (1983b) reported leaf tip concentrations for Typha latifolia that ranged up to 91 ug/g and root levels ranging up to 388 ug/g in plants from a metal enriched area. Nickel leaf concentrations of 467 ug/g have been obtained without toxic symptoms, indicating an internal exclusionary mechanism (Taylor and Crowder 1983a). Below-ground tissue concentrations have been correlated with soil/sediment concentrations, but no correlation between leaf and soil levels have been found (Taylor and Crowder 1983b).

9.3.3.2 Effects of Salix boothii in Remediating AMD

Like Carex rostrata the metal concentrations in Salix boothii were generally higher in below-ground compared to above-ground plant tissue (Table 9-6). This pattern occurred at both the background and Swamp Gulch wetlands and for all elements, with the possible exception of Zn. Zinc appeared to be more concentrated in the leaves/stems than the roots/rhizomes, especially in the Swamp Gulch wetland.

Aluminum was absorbed by Salix boothii and concentrated in the below-ground tissue as opposed to the leaves and stems. The level of Al in the below-ground tissue from the Swamp Gulch wetland was substantially higher (3.4 times) than the tissue concentration from the background site (Table 9-6). However, above-ground tissue concentrations were similar between wetlands. The ability of Salix boothii to absorb and store Al may have

Table 9-6. Concentration (ug/g) of elements in Salix boothii.

Element	Matrix ¹	Background Wetland (n=2)		Swamp Gulch Wetland (n=6)	
		Mean	Range	Mean	Range
Al	AG	51	34-68	50	35-74
	BG	190	110-270	655	170-1640
As	AG	0.35 ²	0.35-0.35	0.35	0.35-0.35
	BG	2.9	2.3-3.4	3.6	0.35-10.0
Cd	AG	0.6	0.5-0.6	5.6	1.0-9.9
	BG	0.6	0.5-0.7	9.1	2.3-17
Ca	AG	6680	6560-6810	6817	5040-7750
	BG	8580	7590-9570	4535	3240-4860
Cu	AG	0.35 ²	0.35-0.35	1.3	0.35-6.0
	BG	11	8.7-13	169	0.35-390
Fe	AG	1535	890-2180	132	59-420
	BG	3000	1830-4290	18418	4470-33000
Pb	AG	0.7 ²	0.7-0.7	1.7	0.7-3.8
	BG	3.9	1.7-6.0	112	9.2-350
Mn	AG	110	99-120	257	150-500
	BG	125	120-130	456	63-1380
Ni	AG	0.7 ²	0.7-0.7	0.9	0.7-1.6
	BG	0.7	0.7-0.7	1.6	0.7-2.7
Zn	AG	130	110-150	388	210-500
	BG	105	100-110	278	160-420

¹Plant material: AG=above-ground, BG=below-ground

²Values below the instrument detection limit were estimated by multiplying the instrument detection limit value by 0.7.

contributed to the decrease in Al measured in water leaving the Swamp Gulch wetland (Table 9-4).

Arsenic tissue concentrations were similar between the background and the Swamp Gulch wetland. This was expected since As was not elevated in the influent AMD. In the above-ground tissue, As was well below the phytotoxic range (5-20 ug/g) reported for terrestrial plants (EPA 1987a). Arsenic in the below-ground tissues ranged from 2.9 to 3.6 ug/g, indicating that As accumulates in these materials over time.

Salix boothii tended to accumulate Cd, despite the fact that it was present at a low concentration in the influent water (Table 9-4). The level of Cd in the AMD entering the wetland was 0.030 mg/L compared to 0.001 mg/L in the non-enriched water above the Carbonate Mine site. This increase in dissolved Cd was enough to increase tissue levels more than nine times background (Table 9-6). The absorbance of Cd by Salix boothii may have contributed to the reduced level of Cd measured in water leaving the Swamp gulch wetland at station F-1. Based on the tolerable level (10 ug/g) for terrestrial plants (EPA 1987a), it is reasonable to assume that Salix boothii is not experiencing toxic effects from Cd.

Calcium behaved differently in Salix boothii than in Carex rostrata. For Carex rostrata, Ca levels were lower in plant tissues from the Swamp Gulch site compared to the background site. For Salix boothii, this pattern only occurred for the below-ground tissues. The concentration of Ca was 8580 ug/g in the roots at the background site compared with 4535 ug/g at the Swamp Gulch wetland. Calcium levels in the above-ground tissues were similar at the background (6680 ug/g) and the Swamp Gulch (6817 ug/g) wetlands, respectively.

Copper tissue concentrations were higher in above- than below-ground material and were also higher at the Swamp Gulch wetland compared to the background wetland (Table 9-6). The concentration of Cu was 3.6 (above-ground tissue) and 15.3 (below-ground) times greater in tissues from the Swamp Gulch wetland than the background wetland. Salix boothii probably contributed to the removal of dissolved Cu from the AMD as it flowed through the wetland. Above-ground tissue levels for Salix boothii were well below the phytotoxic level of 20 ug/g for terrestrial plants (EPA 1987b).

Iron concentrations were generally lower in the tissues of Salix boothii than in Carex rostrata. In the above-ground tissues for instance, the Fe level in the Swamp Gulch wetland was 132 ug/g, which was notably lower than the 1532 ug/g in tissue from the background wetland on Hardscrabble Creek. Iron concentrations in below-ground tissues were higher in the Swamp Gulch wetland compared to background tissue levels.

Lead tissue levels were lower in Salix boothii than in Carex rostrata. The Pb was more concentrated in below- than above-ground tissue and more concentrated in tissues from the Swamp Gulch wetland than the background site. Compared to phytotoxic values for terrestrial plants (>10 ug/g), it is unlikely that Pb caused toxic effects on Salix boothii (EPA 1987a).

Manganese was also distinctly more concentrated in the below- than the above-ground tissue, and was more concentrated in tissue from the Swamp Gulch wetland compared to the background site. These patterns differ from those of Carex rostrata where Mn levels were highest in the leaves and similar between sites. The data suggests that Salix boothii may have a greater affinity of Mn than Carex rostrata. Despite the absorption of Mn by Salix boothii, the concentration of Mn in the water was not reduced. The influent water value was 3.74 mg/L and the value at station F-1 was 4.30 mg/L (Table 9-4).

Salix boothii did not appear to accumulate Ni to a notable degree. Tissue levels at the background site were below the instrument detection limit of 1.0 ug/g, so values have been estimated (Table 9-6). Tissue concentrations at the Swamp Gulch site were not phytotoxic.

Salix boothii responded to higher Zn levels in AMD at the Swamp Gulch wetland by absorbing more. Tissue levels of Zn were higher at the Swamp Gulch wetland than the background site. Salix boothii tended to store this element in leaves and stems rather than in roots. At the background wetland, Zn levels were 130 ug/g in the above-ground tissue and 105 ug/g in the below-ground tissue. Likewise, Zn levels at the Swamp Gulch wetland were 388 and 278 ug/g in the above- compared to below-ground tissues, respectively. The Zn levels in these wetlands were well below the 500 ug/g phytotoxic level for terrestrial plants recommended for the Helena Valley (EPA 1987a).

9.3.3.3 Effects of Betula glandulosa in Remediating AMD

Betula glandulosa tended to absorb and accumulate metals in much the same way as Salix boothii. In general, tissue concentrations for the analyzed metals were higher in the below-ground than in the above-ground plant material. This was true for both wetlands studied and for all elements, with the exception of Mn and Zn (Table 9-7). These patterns were similar for the accumulation of Zn in Salix boothii and the accumulation of Mn in Carex rostrata.

With the exception of Al and Ca, plant tissue metal concentrations were higher in the Swamp Gulch wetland than at the background site. Aluminum levels were similar between the wetlands, while Ca values were lowest in samples from the Swamp Gulch wetland. The relatively low tissue metal levels make phytotoxicity of Betula glandulosa unlikely.

9.3.3.4 Effects of Bryophytes in Remediating AMD

Metal levels were determined for two mosses, Isopterygium pulchellum and Sphagnum tenellum. Isopterygium pulchellum was

Table 9-7. Concentration (ug/g) of elements in Betula glandulosa.

Element	Matrix ¹	Background Mean (n=2)	Wetland Range	Swamp Gulch Mean (n=6)	Wetland Range
Al	AG	35	29-41	28	23-31
	BG	141	12-270	151	67-250
As	AG	0.35 ²	0.35-0.35	0.6	0.35-1.6
	BG	0.35	0.35-0.35	0.7	0.35-1.6
Cd	AG	0.35 ²	0.35-0.35	0.5	0.35-0.7
	BG	0.35	0.35-0.35	1.4	0.5-2.0
Ca	AG	6835	5670-8000	3470	2730-4320
	BG	2150	1600-2700	1808	1070-3060
Cu	AG	0.35 ²	0.35-0.35	2.8	0.35-8.8
	BG	7.1	0.35-14	15	0.35-48
Fe	AG	73	57-89	158	38-630
	BG	205	120-290	1417	3510-26200
Pb	AG	0.7	0.7-0.7	1.4	0.7-2.4
	BG	5.5	5.1-26	19	4.2-49
Mn	AG	81	76-86	452	120-850
	BG	39	11-67	186	110-350
Ni	AG	0.7 ²	0.7-0.7	1.1	0.7-1.8
	BG	0.7	0.7-0.7	0.7	0.7-0.7
Zn	AG	185	160-210	229	71-580
	BG	56	30-82	147	84-320

¹Plant material: AG=above-ground, BG=below-ground

²Values below the instrument detection limit were estimated by multiplying the instrument detection limit value by 0.7.

present at both wetlands, while Sphagnum tenellum was found only at the Swamp Gulch site.

Table 9-8 presents tissue concentration data for various bryophyte species in metal-enriched and non-enriched environments. The range of values for a particular element varies widely due to the different species analyzed and because of the different environments in which they were found. For some of the elements (Fe, Pb, Mn, Ni and Zn), the range of values for the non-enriched areas overlaps the range for the enriched Table

Table 9-8. Accumulation of elements by bryophytes in various environments.

Reference	Chemical Environment	Concentration (ug/g)									
		Al	Cd	Ca	Cu	Fe	Pb	Mn	Ni	Zn	
Lee et al. 1984 ¹	Enriched		3-259		1.0-2.0	42-646	19-401	2-8	5-28	526-3140	
	Non-enriched		1		1	97	1	4	5	80	
Gignac & Beckett 1986 ²	Enriched	1000-10000		2000-12000	<100-1260	1000-18000		10-190	40-870		
Leland et al. 1979 ³	Urban		1-2		13-15	158-2835	5-309	39-389		26-132	
Percy 1983 ⁴	Non-enriched		0.2		3	353	9	245	2	24	
Wehr & Whitten 1983 ⁵	Enriched		1-31			2190-39000				445-23000	
Wieder et al. 1985 ⁶	Enriched	2806-2903				5619-36636		291-412			
Range for non-enriched environments		---	0.2-1	---	1-3	97-353	1-9	4-245	2-5	24-80	
Range for enriched environments		1000-10000	3-259	2000-12000	13-1260	42-39000	5-401	2-412	5-870	26-23000	

Four species studies: ¹*Pohlia wahlenbergii*, *Leptobryum pyriforme*, *Philonotis fontana*, *Drepanocladus aduncus*. Water naturally enriched with Zn, Pb and Cd.
²*Sphagnum* peat from natural wetlands enriched by fallout from a Ni smelter.
³*Sphagnum* sp. and feather moss from a natural wetland in an urban area.
⁴*Sphagnum magellanicum* from a regional survey in the Maritime Provinces, Canada.
⁵*Rhynchostegium riparioides* from natural wetlands downstream from old Pb mine wastes and from industrial sources of Zn pollution.
⁶*Sphagnum* peat in artificial wetland exposed to AMD for 10 months.

areas. Despite this apparent anomaly, these data indicate the general range of element concentrations for mosses.

With the exception of Mn, the element levels in Isopterygium pulchellum (Table 9-9) were within, or close to, the range for other mosses from enriched environments (Table 9-8). Data from previous studies indicate a high level for Mn of 412 ug/g in Sphagnum spp. from an artificial wetland that received AMD (Wieder et al. 1985). In contrast, Isopterygium pulchellum accumulated 1423 ug/g of Mn at the Swamp Gulch wetland. This difference is likely due to the inherent abilities of Isopterygium pulchellum to accumulate more Mn than the Sphagnum species studied by Wieder et al. (1985). However, this difference may also be due, at least in part, to the length of time that the two mosses were exposed to AMD. The Sphagnum moss was exposed over a 10 month period, whereas Isopterygium pulchellum had been growing at the Swamp Gulch site for years. In any case, Isopterygium pulchellum apparently can absorb and store relatively large amounts of Mn.

With the exception of As and Ca, all elements were elevated in Isopterygium pulchellum at the Swamp Gulch wetland compared to background levels (Table 9-9). This result was expected for As since it was not elevated in the influent water (Table 9-4). The pattern for calcium was similar to that for the other plant species since the concentration was lower on the Swamp Gulch compared to the background wetland.

Tissue concentrations of all the elements except Ca were lower in Sphagnum tenellum than in Isopterygium pulchellum at the Swamp Gulch wetland. The concentration of elements in Sphagnum tenellum were within the range or were lower than the range for other mosses from enriched environments (Table 9-8).

9.3.4 Summary of Element Enrichment in Plants

Table 9-10 lists those plant species analyzed at both wetlands and the elements that were enriched above background. Iron, Cu, Pb and Zn appeared to be enriched in all species. The moss, Isopterygium pulchellum, accumulated all elements except Ca. Significant phytotoxicity of any of these elements is probably not occurring at the study site. Plant enrichment of Cd is of particular concern because the threshold level for safe intake of the element by animals is relatively low. This potential problem is discussed in subsequent sections.

Table 9-9. Concentration (ug/g) of elements in bryophytes.

Element	Isopterygium pulchellum			Sphagnum tenellum		
	Background Wetland		Swamp Gulch Wetland Mean (n=4)	Swamp Gulch Wetland		Range
	Mean (n=1)	Range		Mean (n=2)	Range	
Al	760.0	520-3450	1518.0	310.0	220-400	
As	3.6	1.5-6.2	3.9	1.2	1.0-1.4	
Cd	0.7	3.0-11	7.0	1.5	1.0-2.1	
Ca	17500.0	1990-16600	1069.0	5450	3520-7380	
Cu	15.0	0.35-420	132.0	9.0	8.0-10	
Fe	2080.0	16200-61500	29025.0	7440.0	680-14200	
Pb	7.0	12-75	50.0	17.0	4.2-2.1	
Mn	170.0	12-3240	1423.0	190.0	110-270	
Ni	1.2	0.7-11	6.5	4.2	0.7-7.6	
Zn	44.0	44-3020	1074.0	162.0	43-280	

Values below the instrument detection limit were estimated by multiplying the instrument detection limit by 0.7.

Table 9-10. Plant species demonstrating element enrichment (i.e. greater than background levels) at the Swamp Gulch wetland site.

<u>Plant Species</u>	<u>Enriched Elements</u>
<u>Carex rostrata</u>	Al, Cd, Cu, Fe, Pb, Zn
<u>Salix boothii</u>	Al, Cd, Cu, Fe, Pb, Mn, Zn
<u>Betula glandulosa</u>	Cd, Cu, Fe, Mn, Pb, Zn
<u>Isopterygium pulchellum</u>	Al, Cd, Cu, Fe, Pb, Mn, Ni, Zn

9.3.5 Vegetation Element Levels and Wildlife Hazards

Concern has been expressed that the construction of artificial wetlands for AMD abatement may result in high levels of metals in the vegetation and that this would contaminate or poison the wildlife resource. The following sections address this concern by comparing animal hazard criteria levels with plant tissue data demonstrating elevated levels of the target elements.

9.3.5.1 Dietary Hazard Levels for Animals

There is a general lack of data on hazardous metal intake levels for wildlife species. However, there is considerable data for domestic animals. Table 9-11 lists the maximum tolerable levels of minerals in diets of cattle, horses and domestic rabbits. When these levels are fed for a limited period, they will not impair animal performance and should not produce unsafe levels in human food derived from these animals. These threshold values were generated from a variety of studies in which graded levels of the elements were ingested and the specific affects examined. Highly soluble and purified forms of these elements were generally used in the studies. This practice tended to bias the threshold levels low. In other words, a particular concentration of a metal in the diet may exceed the threshold level but not be dangerous. In addition, adverse physiological effects may not occur at a particular threshold level if the element has been complexed into plant matter during growth and is not digested by the animal. Animal tolerance to chemical elements will vary with age and physiological condition. For the purpose of discussion, these data (Table 9-11) will be used as a general guide for potential contamination of wildlife species utilizing the Swamp Gulch wetland.

Table 9-11. Maximum tolerable levels¹ of dietary minerals for domestic animals (National Academy of Sciences 1980).

Element	Species		
	Cattle	Horse	Rabbit
Al	1000	(200)	(200)
As	50	(150)	50
Cd	0.5	(0.5)	(0.5)
Ca%	2	2	2
Cu	100	800	200
Fe	1000	(500)	(500)
Pb	30	30	(30)
Mn	1000	(400)	(400)
Ni	50	(50)	(50)
Zn	500	(500)	(500)

¹ All values are in ug/g unless otherwise specified. Values in parentheses were derived by extrapolation.

9.3.5.2 Wildlife Observed and at Risk

Wildlife observations were recorded during each field trip. Voles (Microtus spp. and possibly Clethrionomys gapperi) were the animals most often observed. They were generally seen among Carex rostrata where the soil was saturated or nearly so. One cottontail rabbit (Sylvilagus nuttalli) was seen in the center of the site. The only ungulates actually observed were mule deer (Odocoileus hemionus). A total of six adult deer were seen during the study period while a fawn and doe were observed several times. Tracks of moose (Alces) and possibly elk (Cervus canadensis) were seen twice. Their use of the area was minimal. Evidence of beaver (Castor canadensis) use of the site was low. Beaver tracks were observed only once and freshly chewed willow stems were seen on several trips. Beaver were probably moving through the area but they were definitely not residents of the site. Brook trout (Salvelinus fontinalis) were often in the larger channels in the southeast portion of the wetland study area. An approximately eight inch brook trout was seen in a channel near station AAA-3. Only very small trout were observed in the Blackfoot River. Numerous frogs were observed throughout the Swamp Gulch wetland. Coyote (Canis latrans) tracks were seen on only one occasion. Very few passerine birds were observed. A redtail hawk (Buteo jamaicensis) and presumably a great horned owl (Bubo virginianus) were observed once.

Animal species most likely to be poisoned by heavy metals are those whose home ranges encompass a metalliferous environment. In the case of the Swamp Gulch wetland, these would be

primarily herbivores such as voles, muskrats (Ondatra zibethica) and beaver, and carnivores such as weasel (Mustela frenata) and mink (Mustela vison). Other possible residents of the study site that could be at risk are shrews (Sorex spp.) and the skunk (Mephitis). Mammals and birds that feed over a large area are not likely to be poisoned by occasionally consuming vegetation or contaminated animals from the Swamp Gulch wetland. These include ungulates such as elk, moose and mule deer, and predators/scavengers such as coyote, fox and various raptors. This discussion will focus on those species with small home ranges, meaning those more likely to consume large quantities of vegetation or other animals from the Swamp Gulch wetland.

9.3.5.3 Hazards to Wildlife at the Swamp Gulch Wetland

Cadmium appears to be the only element present in the macrophytes (Tables 9-3, 9-6 and 9-7) at levels that could be toxic to animals. The bryophytes contain metal levels that exceed maximum dietary intakes, however, these plants are not consumed in quantities that could cause metal poisoning. Of the macrophytes, Salix boothii had elevated Cd levels on the Swamp Gulch wetland that greatly exceeded the maximum dietary intake level. The Cd level in the above-ground tissues was 5.6 ug/g which was significantly greater than the dietary tolerable level of 0.5 ug/g (Table 9-11). Beavers consume large quantities of Salix boothii, so could be at risk for Cd poisoning. If beaver have been poisoned by consuming Salix boothii, it would help explain the absence of resident beavers at the site. Beaver are currently active in areas upstream and downstream of the Swamp Gulch wetland and were once very active at the site.

In summary, Salix boothii on the Swamp Gulch site had a level of Cd that exceeded the maximum tolerable dietary intake for domestic animals. Beaver appear to be at particular risk from Cd poisoning because of their heavy utilization this shrub. Other animals, such as voles, may also be at risk. If these small rodents are accumulating high levels of Cd then the animals that prey on them (weasel, mink, owl) may also be experiencing toxic effects. It is less likely that infrequent visitors to the Swamp Gulch wetland such as elk, deer, moose, coyotes, foxes and birds would be poisoned.

At this point, it is unknown whether true metallophytes exist at the Swamp Gulch study site. If these types of plant species are present, certain animals may be at special risk if they consume these plants. Future work at this site should include an examination of metal levels in tissues of the common herbivore(s). These data would help determine the extent to which certain metals are present in the food chain.

10.0 MICROORGANISMS AND THE IMMOBILIZATION OF HEAVY METALS IN WETLANDS

10.1 BACKGROUND

This portion of the study characterized microbial communities and conditions associated with AMD in constructed and natural wetlands using a new method for measuring the populations of microorganisms based on their unique biochemical components. MSI Detoxification, Inc. (MDI) uses this technology to characterize hazardous waste sites and proposed its implementation in this study. This new method allows the analysis of mixed microbial populations found in natural environments and does not require cultivation of organisms in the laboratory. Thus, it is a powerful tool which avoids isolation of microorganisms by traditional cultivation techniques, and therefore, may be more representative of the microorganisms present in situ. As a part of the study of the efficacy of natural wetlands in the remediation of acid mine drainage, it was determined that the role of microorganisms, specifically sulfate reducing bacteria, be investigated in the unique circumstances presented by the Swamp Gulch natural wetland in Lincoln, Montana. A reference for this wetland was provided by the Sand Coulee wetland, a constructed wetland near Tracy, Montana.

The hypothesis for the perceived success of natural wetlands in remediating AMD considers that microorganisms have the ability to carry out the reduction of in-flowing sulfate. This, in turn, raises the pH of the system and produces sulfide. The sulfide may be involved in reductions of ferric iron (Fe^{3+}) to ferrous iron (Fe^{2+}) and subsequent immobilization, possibly as inorganic or organic sulfides. If the process (or the process initiator) is microbial rather than chemical, then sterilized soil samples should not cause the pH of AMD to rise after incubation under anaerobic conditions.

To test this hypothesis in the Swamp Gulch environment, experiments were performed under which pH, sulfate, iron, and microbial biomass would be measured over time and related to AMD remediation. The experimental approach was divided into three Phases. Under Phase I, the microbial component of the Swamp Gulch AMD remediation activity was investigated through a comparison of remediation capacities in sterilized versus untreated core materials (soils, organic detritus, inorganic materials). Phase II sought to verify the presumed microbial activities observed under Phase I by following pH, sulfate, and iron in wetland core materials in a controlled greenhouse environment. Under Phase II, materials from the Swamp Gulch natural wetland were compared with sediment core materials taken from the Sand Coulee constructed wetland. Phase III focused on the microbial component of the wetlands and compared biomass and

(inferred) microbial diversity in field samples extracted from a natural, constructed and control wetland (Hardscrabble Creek near Lincoln, Montana).

The characterization of the major types of microorganisms in artificial and natural wetlands and their correlation with effectiveness of AMD remediation are intended to allow us to develop rapid and effective methods to diagnose and renovate malfunctioning wetlands.

10.2 PHASE I: SWAMP GULCH WETLAND BUFFERING CAPACITY AND THE EFFECT OF SOIL STERILIZATION ON WETLAND PH

10.2.1 Introduction

A preliminary experiment to compare the abilities of Swamp Gulch wetland soil samples to remediate AMD under various conditions was performed. The purpose of this experiment was two-fold: 1) to gain insight regarding the buffering capacity of Swamp Gulch soil on exposure to AMD, and 2) to determine if there was evidence that microorganisms beneficially raise the pH in Swamp Gulch wetland soil.

10.2.2 Technical Approach

Soil samples from the Swamp Gulch wetland were incubated with AMD under aerobic and anaerobic conditions. Initial pH values of the soil-AMD mixture and those of the same suspensions after incubation for nine days were compared.

10.2.3 Materials and Methods

10.2.3.1 Soil Sampling

Soil samples (0.5 and 2.0 g) were obtained from cores taken from the Swamp Gulch wetland (Figure 10-1). Each sample was placed in a plastic "Whirl-pak" bag and transported to the laboratory where it was transferred to a Waring Blender and homogenized for a total of 60 seconds. Soil samples (0.5 g and 2.0 g) were weighed into tared 30 ml beakers (titration experiment) or Hungate roll-tubes (microbiological experiment), respectively.

10.2.3.2 Titration Experiment

To determine soil buffering capacity, duplicate 0.5 g soil samples suspended in 10 ml of distilled water were titrated with AMD. Readings of pH were made for six minutes or until a stable value was obtained (< ten minutes). Similarly, AMD was titrated into distilled water for comparative purposes. AMD titrated into distilled water was not replicated. All samples were

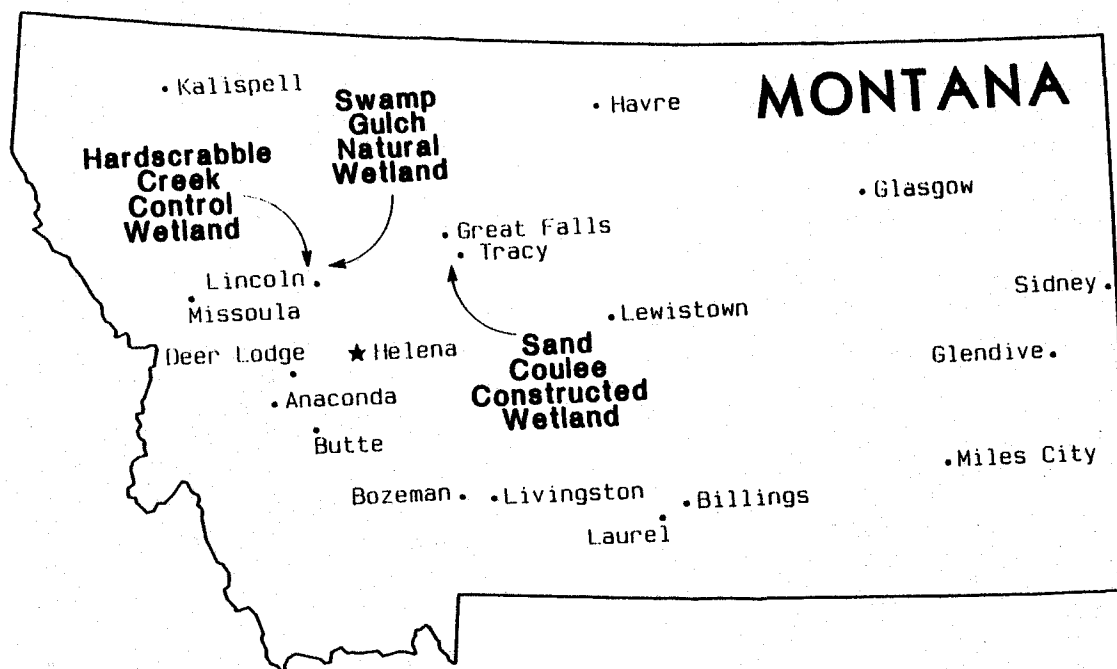


Figure 10-1. Location of wetland sites.

continuously mixed using a magnetic stirring device during the titration.

10.2.3.3 Microbiological Experiment

In the microbiological study all sampling was carried out in triplicate and three treatments were considered: a sterilized control, an aerobic and an anaerobic treatment. All samples were incubated under ambient laboratory temperature conditions for a period of nine days. For the sterilized control, soil samples (2.0 g) suspended in 14 ml water or 14 ml AMD were sterilized on three consecutive days by autoclaving for 15 minutes at 15 psi. Anaerobic treatment samples were gassed with O_2 -free N_2 and stoppered with Hungate stoppers. Aerobic treatment samples were incubated in 125 ml Erlenmeyer flasks with loose fitting caps and incubated with vigorous shaking. Readings of pH were made initially and after the incubation period.

After incubation and pH measurement, triplicate samples of each treatment were pooled to provide sufficient sample for iron analysis. The pooled samples were centrifuged at $10,000 \times G$ for ten minutes and the supernatant fraction decanted. This

fraction was acidified with concentrated HCl and total dissolved iron determined by the phenanthroline colorimetric method (APHA 1985, Iron Determination, pp. 215-220).

10.2.4 Results and Discussion

10.2.4.1 Titration Experiment

Figure 10-2 depicts the titration of wetland soil homogenate and distilled water. After a volume of AMD was added to the stirred soil suspension, the pH fell to a minimum of 4.5 during the six minute measurement period. This indicates that the soil has a capacity to adsorb acidity independent of microbial involvement, since the experiment was conducted on time scales too short for significant microbial activity.

10.2.4.2 Microbiological Experiment

The incubation under anaerobic conditions of AMD soil from the Swamp Gulch wetland raised the pH of the AMD by 0.72 units (Table 10-1). When the soil and AMD were sterilized, no such

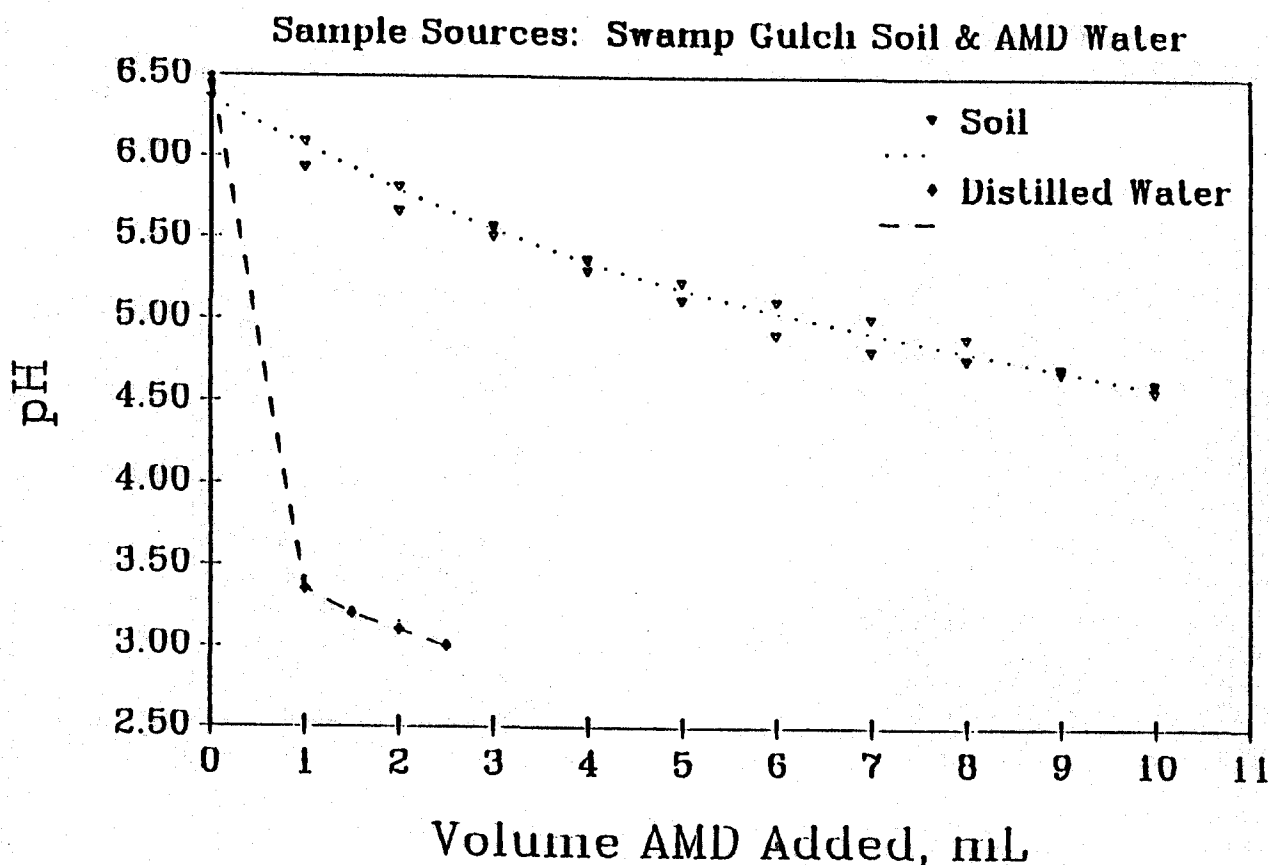


Figure 10-2. Titration of Swamp Gulch soil with AMD.

Table 10-1. pH and iron values¹ for AMD incubated under various treatment conditions after a period of nine days.

<u>Treatment</u>	<u>Pre Incubation pH</u>	<u>Post Incubation pH</u>	<u>Net Difference</u>	<u>Total Dissolved Iron, mg/L</u>
Aerobic	7.08±0.03a	5.17±0.13c	-1.92±0.10a	162.9±2.5ab
Anaerobic	6.28±0.03b	7.00±0.05a	+0.72±0.06b	20.7±0.0 ^b
Sterile Control	6.35±0.05b	6.37±0.10b	+0.02±0.06c ³	172.8±5.1a

¹ Means ± standard deviation; n = 3 for pH data; n = 2 for total dissolved iron data.

² Treatment means (within columns) are significantly different (P=0.01) if followed by a different letter.

³ The sterile control pH means from preincubation to postincubation were not significantly different from one another.

increase in pH took place. Anaerobic conditions were necessary for this process. When AMD was incubated aerobically, the pH fell by 1.92 units and the soil became bright orange, presumably due to the production of iron oxides. In all anaerobic incubations the soil maintained its dark brown/black color which would be expected if metal sulfides were being produced. Comparison of iron concentrations in the presence and absence of oxygen indicate that there was much less iron in the anaerobically treated sample. Presumably this anaerobic response is the consequence of microbial sulfate reduction to sulfide followed by the precipitation of iron sulfide, which is black in color. To verify that sulfides were present the soil was treated with 2N HCl and an H₂S odor was detected.

The Swamp Gulch wetland soil was able to raise the pH of the applied AMD from 2.5 to about 5.0 in the absence of microbiological contribution (titration experiment). As more AMD was applied to the soil, this dropped to 4.5 (Figure 10-2). Because the pH scale is logarithmic, this represents a 50 to 100-fold increase in the ability of the soil to neutralize acidity (as hydrogen ions). Aerobic incubation resulted in a final pH (near 5) which was similar to that of the soil titration experiment. However, anaerobic incubation raised the pH to 7.0 (4.5 pH units) indicating that wetland soil combined with its indigenous anaerobic microorganisms was able to neutralize AMD acidity by a factor of greater than 10,000-fold.

These findings support the hypothesis that sulfate-reducing bacteria play a notable role in the remediation of AMD. Further evidence in support of the involvement of sulfate-reducing bacteria comes from the phospholipid fatty acid analyses of core samples taken from the same area of Swamp Gulch wetland (Phase III).

10.3 PHASE II: COMPARISON OF NATURAL AND CONSTRUCTED WETLAND SOILS IN THE REMOVAL OF ACIDITY, SULFATE AND IRON

10.3.1 Introduction

Biological oxidation of reduced sulfur due to the activities of sulfur oxidizing bacterial action on pyritic ores is the primary cause in the formation of AMD waste water problems. The resulting products in the waste are acidity, sulfate ion and iron. The sulfate-reducing bacteria remove sulfate by reducing it to sulfide, which can cause the co-precipitation of heavy metals as sulfides. The objective of these experiments was to compare acidity, sulfate and iron removal in a natural and an artificial wetland system. In addition these studies were carried using samples of sufficient size to minimize inhomogeneities which would be present in small samples.

Iron, sulfate and pH measurements were included in the experimental protocol, as microbial sulfate reduction is the presumed basis for AMD remediation through immobilization of heavy metals. Iron was chosen as an indicator metal due to its ease of determination and its ubiquitous presence in the wetlands under study.

10.3.2 Technical Approach

The study was carried out in a greenhouse at the Montana State University Plant Growth Center. Three of the cores were obtained from the Swamp Gulch natural wetland near Lincoln, Montana (Figure 10-1). These were designated SG-1, SG-2, and SG-3 for this study. The other three cores were obtained from the Sand Coulee constructed wetland large cell near Tracy, Montana (Figure 10-1). These cores were designated SC-V, SC-VN, and SC-CH.

The cores received AMD that had been collected at the Swamp Gulch site. AMD was eluted through all of the cores for the duration of the experimental study. The effluent (eluate) as well as the influent AMD (eluent) was analyzed for temperature and pH several times a week. Sulfate and iron concentrations were determined periodically over the course of the study.

Direct microbiological parameter measurements were not made under this Phase of the study. Microbial diversity and biomass analyses comprised phase III of the investigation.

10.3.3 Materials and Methods

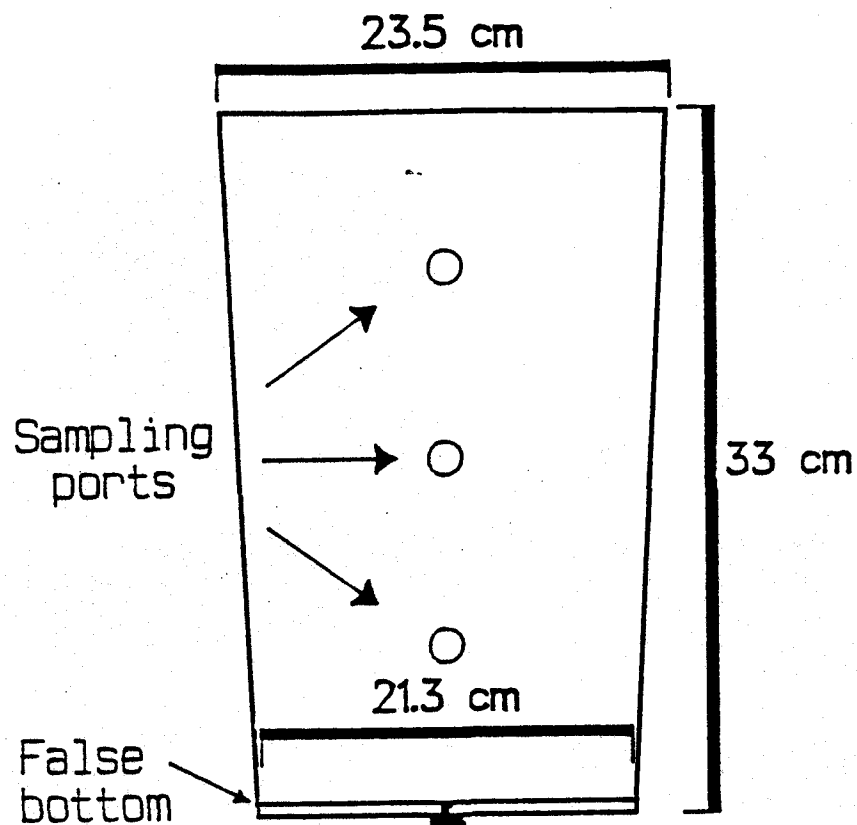
10.3.3.1 Field sampling

A site was selected at the Swamp Gulch wetland and was sampled to provide sediment cores for the greenhouse study. The selected site was between sites C3 and C2 (see Figure 4-1). It was assumed that the remediation capacity of the Swamp Gulch wetland was not saturated at this field location. Site vegetation was dominated by Carex with subsidiary Sphagnum moss.

Three sediment samples (designated SG-1, SG-2 and SG-3) were collected on October 22, 1987. Each core vegetation mat was sawn along the sides and bottom with a keyhole saw, and removed from the wetland using a flat-bladed shovel. Each core was laid on its side on the ground surface and trimmed to fit tightly into the sediment container. The containers, five-gallon plastic buckets modified as shown in Figure 10-3, had been acid-washed and disinfected with chlorine bleach solution before they were taken to the field. After pressing the sediment firmly to remove air spaces along the container walls, native water from the sampling trench was poured over each sample so that standing water was visible just below the core top. A five-gallon container of the in situ water was collected so that it would be possible to replenish sample water in case of losses during transport.

Sediment cores were transported to Bozeman the following afternoon. The next morning, samples were sprayed with MAVRIK Aquaflow, a broad-spectrum insecticide (active ingredient = (aRS,2R)-fluvalinate [(RS)-a-cyano-3-phenoxybenzyl-(R)-2-(2-chloro-4-trifluoromethyl) anilino)-3-methylbutanoate]). Fumigation, required of users of the MSU greenhouse facility, was carried out to prevent the spread of disease among experimental plants. It was believed that this treatment did not affect the microbial or chemical integrity of the samples due to its focused surface application.

Samples from the Sand Coulee site near Tracy, Montana were collected on October 30, 1987, and placed into five-gallon containers in an identical manner as were those collected at the Swamp Gulch wetland (Figure 10-3). The three samples were collected about one meter from the middle baffle of the first cell of the wetland, in an area with sporadic Typha plants and pools of standing water on an irregular sediment surface. Such vegetation cover was typical for that cell (Figure 10-4). The absence of a



All joints sealed with silicone.

Three sampling ports on side.

Glass wool covers the inlet of the effluent port.

2.5 cm sand on bottom, below core, above false bottom.

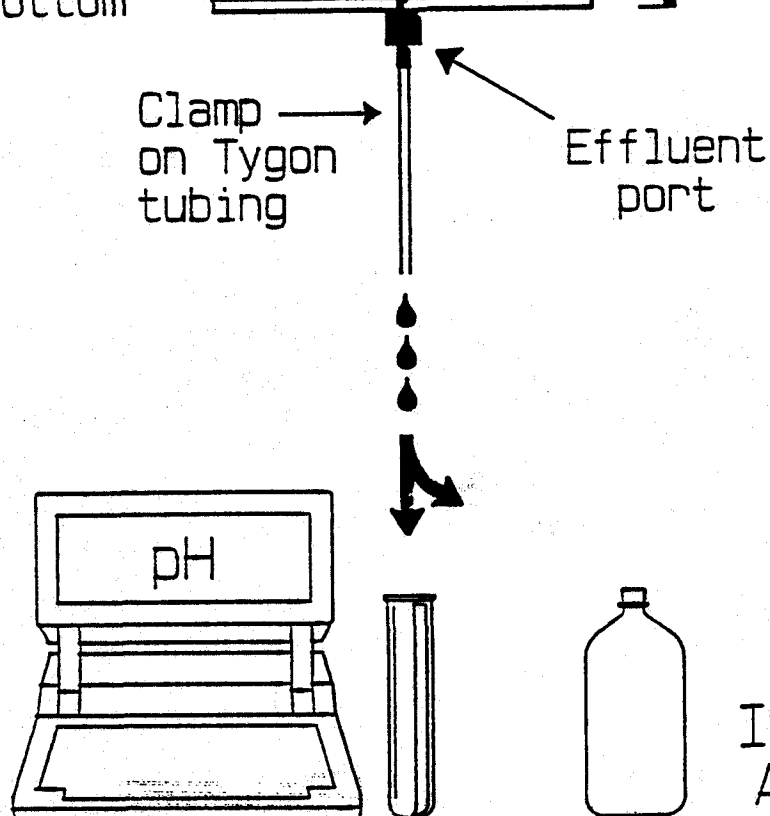


Figure 10-3. Diagram depicting soil core containers used in Phase II greenhouse study.

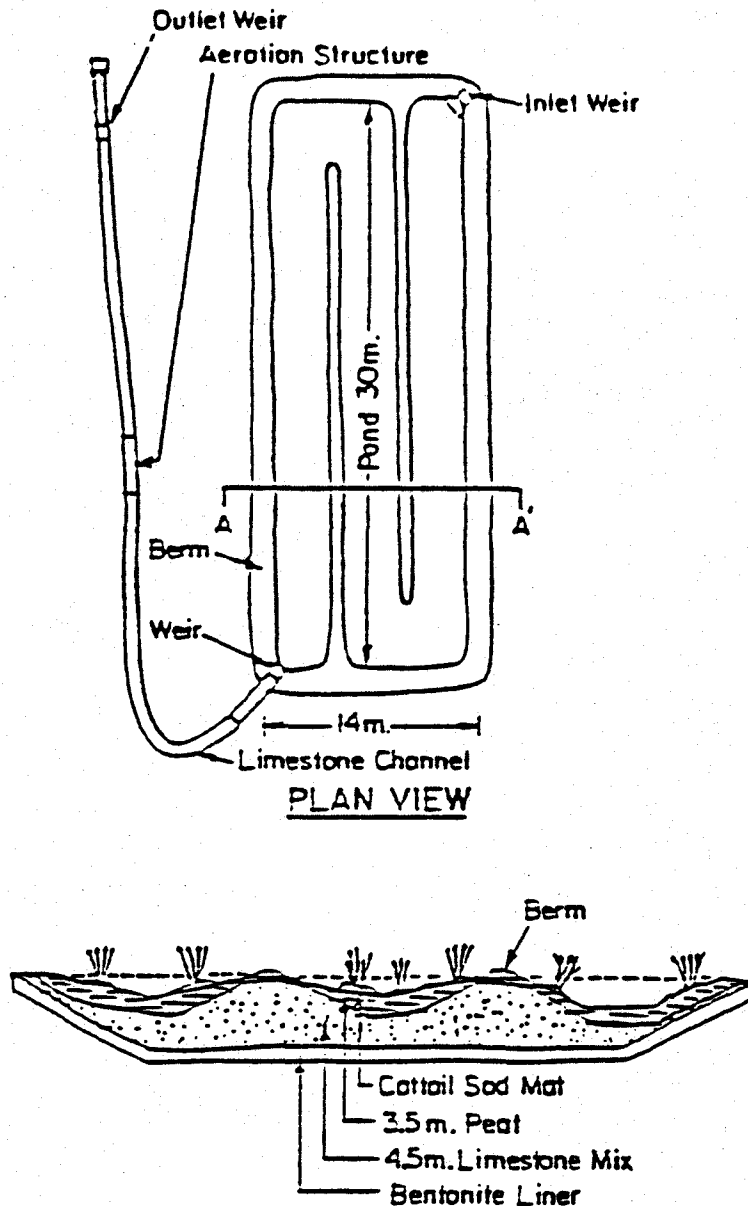


Figure 10-4. Sand Coulee constructed wetland sampling site (Hiel and Kerins 1988).

vegetative mat and the high water level in the artificial wetland precluded removal of a coherent core. Instead, the saturated sediment was shoveled into the container, preserving the depth sequence of the peat. This procedure was deemed acceptable for the Sand Coulee wetland samples because the sediment color and texture were homogeneous throughout the peat profile, and because the sediment was fluid enough to fill the container without forming pockets of air.

Cores were extricated from the Sand Coulee constructed wetland large cell area (designated "SC" cores) for a variety of treatments. The core designated SC-V contained cattails which were cropped for this study. The core designated SC-VN represented a sample void of vegetation during sampling. The core designated SC-CH contained cattails and was treated with chloroform for a period of about two months prior to beginning this study at which time the chloroform addition was halted. The intent in adding chloroform was to chemically sterilize the core. The Sand Coulee constructed wetland cores were treated as replicates and studied in the same manner as were the Swamp Gulch wetland cores.

10.3.3.2 Soil core flow characteristics

The effluent flow rate from each of the cores varied, presumably due to flow characteristics unique to each wetland sample. Flow rates were easily controlled for most cores. However, the effluent flow rate of the SC-V core decreased after a period of time so that less than 200 ml per day would freely flow from the core unless extracted using a vacuum device. This impediment of flow in the SC-V core occurred 26 days into the study and the average overall flow rate dropped to 125 ml/day. Consequently, its behavior during the first 26 days was similar to the five other cores and subsequently was extremely slow (57 ml/day). The flow rate for all other cores ranged from 253 to 291 ml/day.

10.3.3.3 Sampling and preparation of AMD

AMD was collected from the Swamp Gulch natural wetland site by siphoning AMD from a receiving pond, located adjacent to the wetland and on the north side of Montana Highway 200 (see Figure 3-1). The collected AMD was stored in plastic 55-gallon drums that had previously been acid-washed with dilute HCl and rinsed several times with distilled water. The 55-gallon drums were stored closed in the greenhouse.

As AMD water was needed for the experiment, it was siphoned from the 55-gallon drums and filtered through glass wool. The filtered AMD was stored in 5-gallon carboys under glass cover to avoid photooxidation resulting from exposure to sunlight in the greenhouse. This filtered AMD was used as the eluent for both the Swamp Gulch and Sand Coulee wetland cores. Temperature and pH determinations were made on the AMD eluent (influent) and eluate (effluent) collected from each core, every day effluent was sampled.

10.3.3.4 Sulfate and iron determinations

Sulfate and iron concentrations were determined several times during the course of the study period. Dissolved sulfate determinations were made on days 61, 82 and 88 of the study.

Dissolved ferrous and total iron measurements were made on days 42, 61, 82 and 88. The schedule for these analyses was based on an estimate of the percent of one pore volume of water which had been eluted from each of the soil cores. The percent water contained in the soil was determined to be approximately 85% based on percent moisture determinations. Thus, the water contained in each of the core was estimated to be 16.9 L. The core effluent was sampled when approximately 13.5 and 20.3 L had passed through each core and again at the end of the study. Due to the innate low flow rate of the SC-V core it was not possible to obtain iron and sulfate data beyond 12.2 L of flow-through volume.

Sulfate concentrations were determined by ion chromatography (subcontracted to Camas Laboratories, Inc., Missoula, Montana) using standard methods of analysis (APHA 1985, Sulfate Determination, pp. 483-488). Two-hundred mL aliquots of eluate for sulfate analysis were collected from each core and filtered through cellulose acetate membrane filters (nominal exclusion 0.45 μ m). All of these samples were refrigerated (4°C) overnight, packed on ice in a cooler and transported to Missoula, Montana, for chemical analysis. All sulfate analyses were performed within one to three days of receipt of the samples.

Iron analyses were performed using the phenanthroline colorimetric method (APHA 1985, Iron Determination, pp. 215-220). Eluate samples were collected for iron analysis at the same time and in a similar manner as were the eluate samples for sulfate analysis. For dissolved ferrous iron determinations the core eluates and AMD eluent were collected directly into HCl to acidify them immediately and minimize oxidation of the ferrous iron. All dissolved iron samples analyzed were determined on the same day as they were collected; the dissolved ferrous iron samples being determined within nine minutes of collection, and the total dissolved iron samples determined within eight hours of collection. It should be noted that the collection of the samples from the SC-V core, which did not flow freely, was performed under a vacuum pressure.

10.3.4 Results and Discussion

10.3.4.1 Acidity determinations

The temperatures of the influent AMD and the effluents from the cores were generally in the 18-21°C range. The influent AMD pH ranged from 2.17 to 2.70, with a mean and standard deviation of 2.44 ± 0.09 (N=94). This indicated that AMD applied to the various cores remained stable with respect to pH over the study period (Figure 10-5).

The Swamp Gulch wetland core eluates exhibited pH values with nearly identical trends indicating that they were replicates (Figure 10-6). The effluent pH values for the Swamp Gulch wetland

cores ranged from 5.67 to 7.58, with an overall mean and standard deviation of 6.43 ± 0.37 .

The Sand Coulee wetland core eluates, while not true replicates, demonstrated some important trends. Eluate pH values from cores SC-V and SC-VN were similar (Figure 10-7). These pH values dropped at the start of the study, flattening out by 30 days into the experiment and ranged from 2.39 to 4.25, with an overall mean and standard deviation of 3.24 ± 0.45 . The Sand Coulee wetland core which had been treated with chloroform, SC-CH, showed an ascent in its pH for the initial 30 days of the study and a leveling of this pattern subsequently. The SC-CH eluate pH ranged from 4.25 to 5.73 with a mean and standard deviation of 5.12 ± 0.39 .

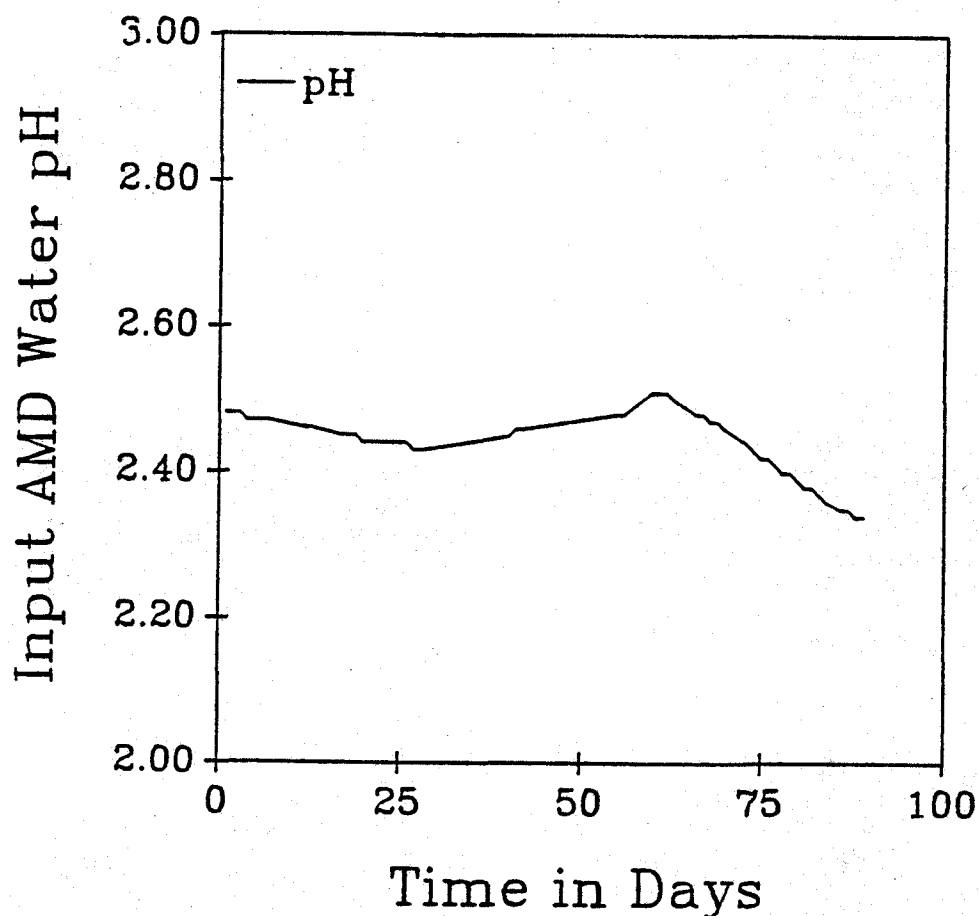


Figure 10-5. Influent (eluent) AMD pH measured over study period.

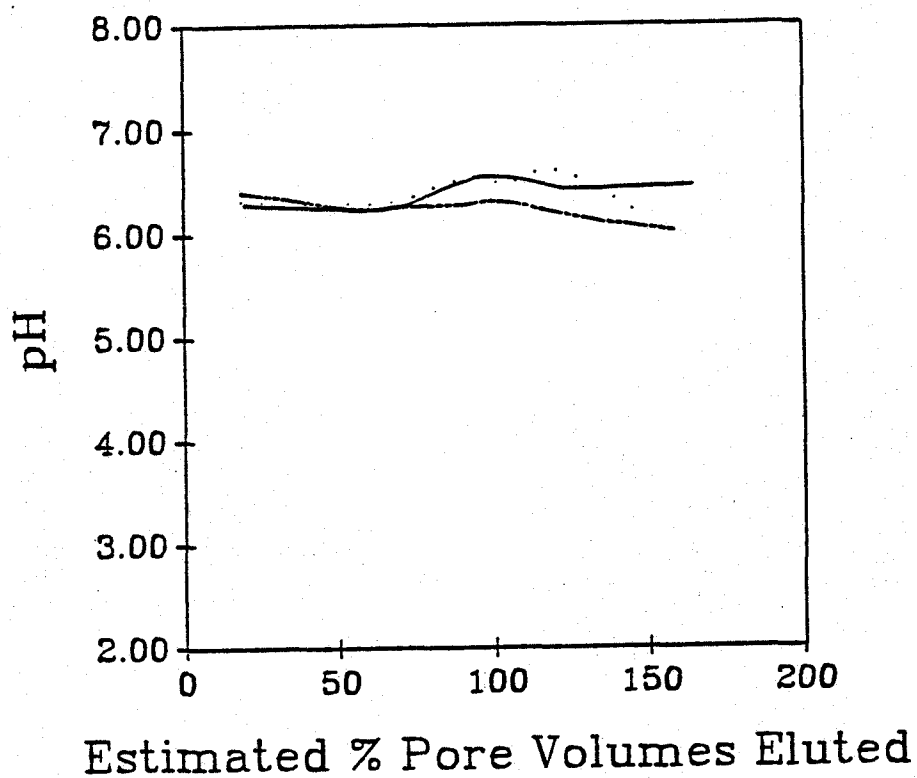
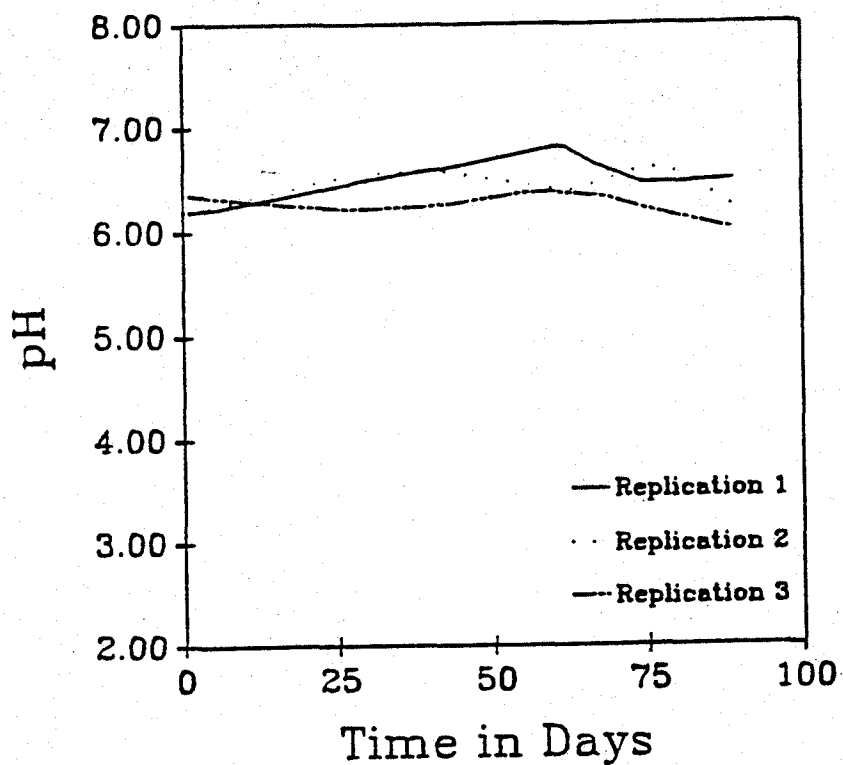


Figure 10-6. Swamp Gulch soil core's effluent (eluate) pH measured over study period.

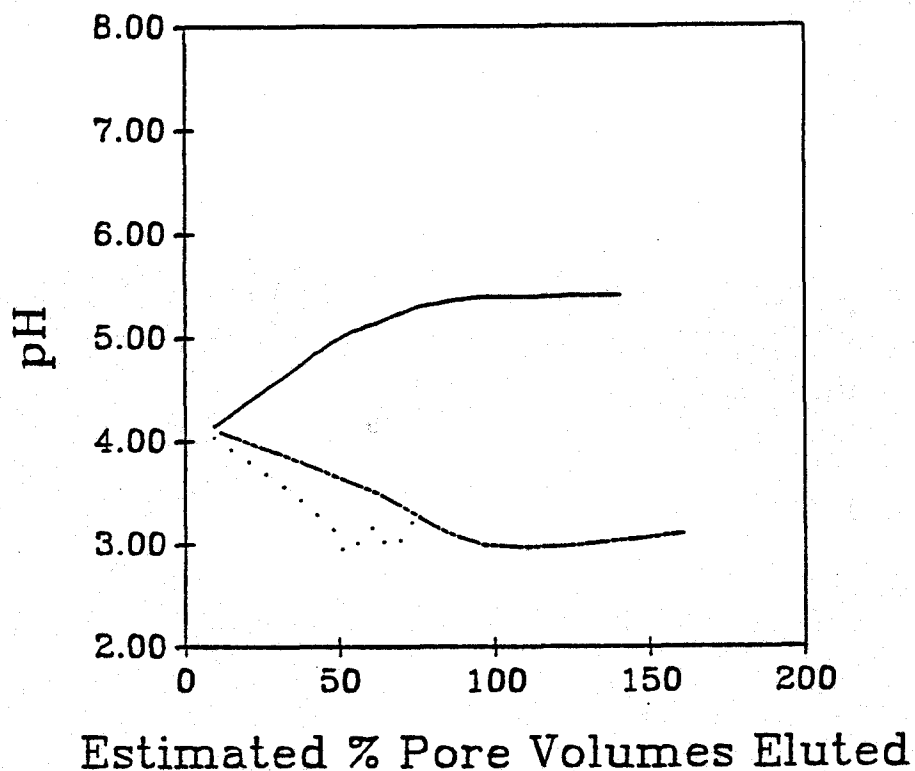
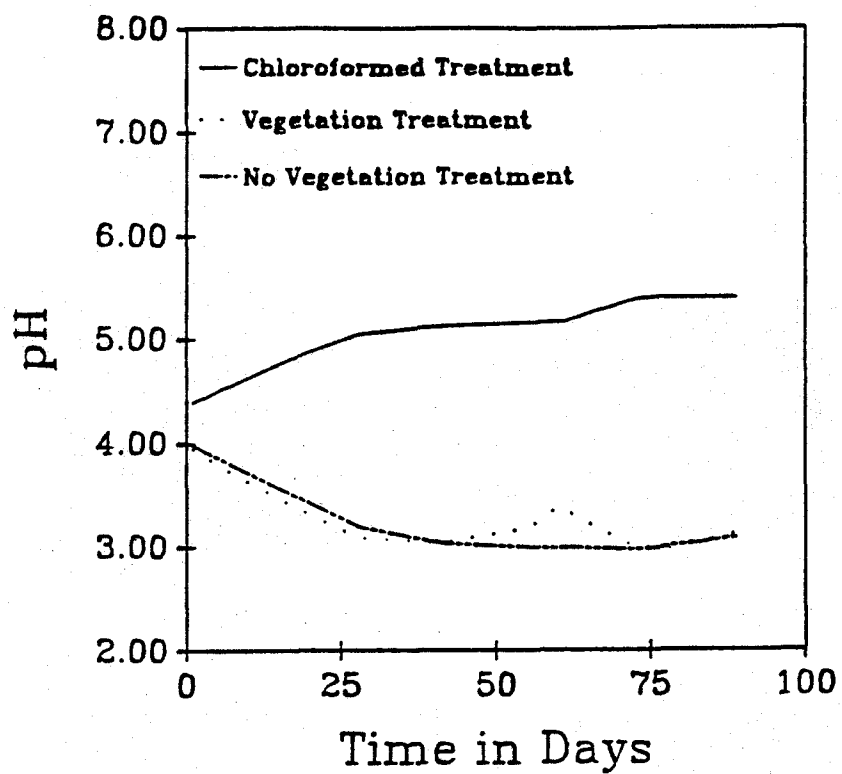


Figure 10-7. Sand Coulee soil core's effluent (eluate) pH measured over study period.

For the SC-V and SC-VN cores for most of the study period, Sand Coulee constructed wetland soil raised pH of applied AMD from 2.5 to about 3.0. It is questionable whether there was any acid neutralizing microbial component in these cores. Consequently, the 0.5 increase in pH may reflect the innate acidity neutralizing capability of the Sand Coulee wetland sediment.

It can be surmised that the application of chloroform to the SC-CH core influenced the rise in pH. Although the eluate pH of the SC-CH treatment never achieved the near-neutral pH (6.43 ± 0.37) typified by the Swamp Gulch wetland cores, it demonstrated remediation of acidity not evident in either of the other Sand Coulee wetland cores.

Two hypotheses can be advanced which would explain this observation. Both of these are based on the principal of microbial selection. In the first, a population of microorganisms arises because the chloroform is lethal to most of the indigenous organisms. A population is selected due to exclusive effects imposed by the chloroform on most of the organisms. As this group of microorganism proliferates, its metabolic activities result in increased pH and amelioration of the acidity. In the second hypothesis, selection occurs through nutritional enrichment of a microbial population able to use the chloroform as a source of carbon and energy. Because they are able to use the chloroform, they come into dominance. Presumably they had the ability to remediate the AMD pH prior to the addition of the chloroform, but were unable to because they were starved for a carbon source.

Because there was no apparent lethal effect of the chloroform in the SC-CH core, and because the cattails in this core did not die, the second hypothesis is more probable. Also, the latter hypothesis is supported by remediation of AMD pH in the Swamp Gulch wetland cores where a high level of detritus from decaying sedges and other plant material was present. Conversely, the Sand Coulee wetland cores lacked conspicuous amounts of detritus which would provide carbon and energy and allow indigenous microbes to proliferate. These findings indicate that the constructed wetland is ineffective in removing acidity, presumably because it lacks sufficient carbon to support indigenous microorganisms. Therefore, it should not be automatically assumed that poor performance in a constructed wetland is due solely to wetland size. This scenario is supported by observations on the phospholipid analysis of the Sand Coulee wetland in situ site (Phase III).

10.3.4.2 Sulfate determinations

The influent AMD applied to the cores exhibited a sulfate concentration of 412 ± 56.1 mg/L, ranging from 336 to 482 mg/L (Figure 10-8). The Swamp Gulch wetland cores are remediating the

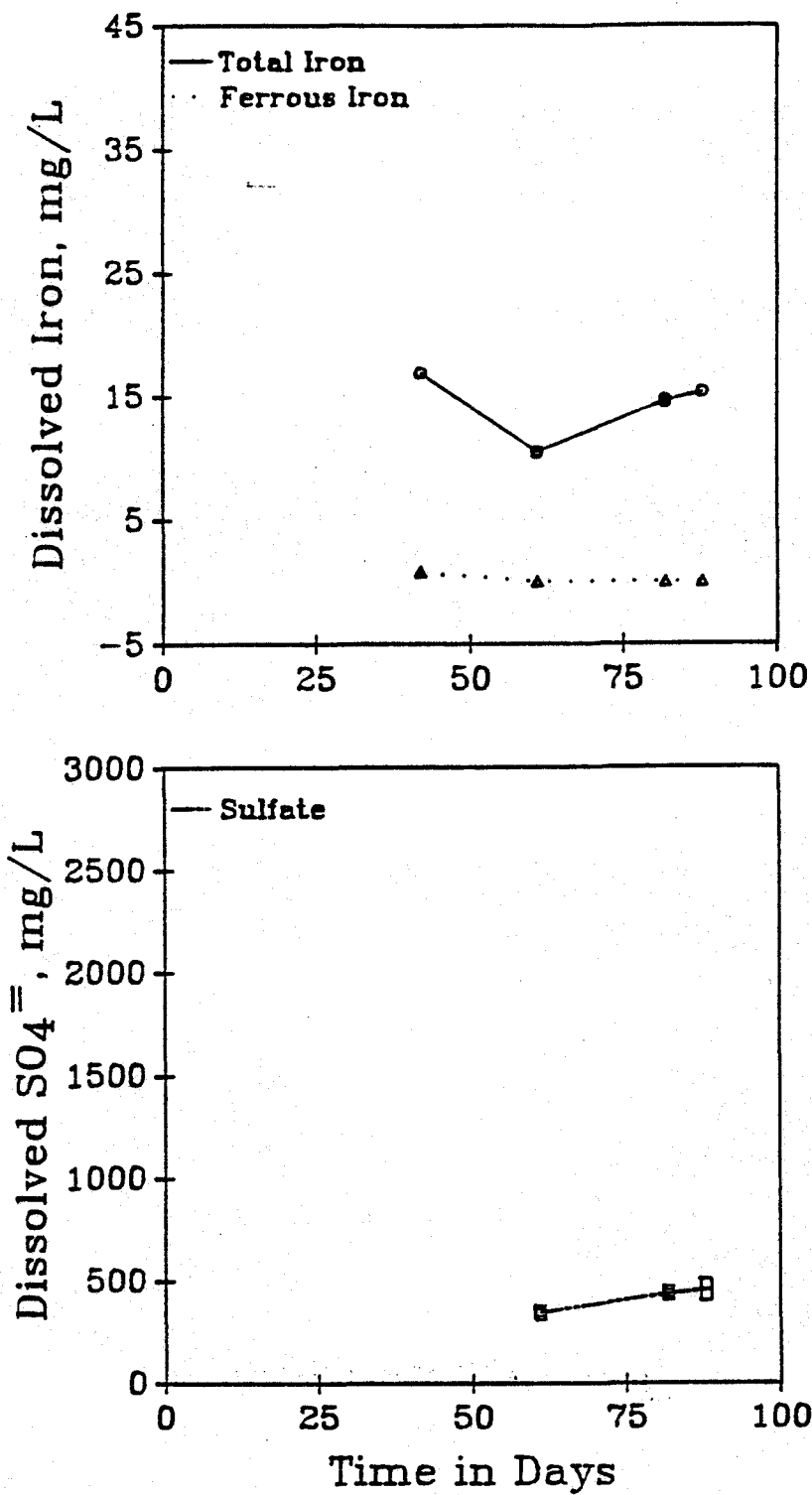


Figure 10-8. Influent (eluent) AMD iron and sulfate concentrations measured over study period.

added sulfate (Figure 10-9). The eluate concentrations for the Swamp Gulch wetland core vary from 0 to 158 mg/L with a mean and standard deviation of 59.5 ± 56.2 mg/L. In general, the remediation was better as the flow rates were slower (Table 10-2).

Sulfate concentrations in the effluent increased as the flow rates were increased. This could be associated with the rate at which indigenous sulfate-reducing bacteria are able to facilitate sulfate reduction to sulfide, where greater retention times provide for greater sulfate reduction. This phenomenon would be particularly important where sulfate-reducing bacteria were either not present, present at low concentrations, or present, but not active (Phase III). For example, in a constructed wetland if flow rates are high and/or channeling is evident, one would expect poor sulfate-reduction performance to occur. Therefore channelization and high flow rates in constructed wetlands should be avoided by implementation of improved engineering design.

Instead of remediating the sulfate concentrations the Sand Coulee wetland cores are releasing sulfate into the effluent (Table 10-3, Figure 10-9). The effluent concentrations for the Sand Coulee wetland cores range from 1385 to 2660 mg/L. Presumably this is because the in situ addition of sulfate at the Sand Coulee wetland site is much higher than that in the Swamp Gulch AMD input water, and it is beginning to be removed as the lower sulfate-containing Swamp Gulch AMD is being applied to the cores (Figure 10-9).

The flow rates for these cores (except SC-V) increased during the later part of the study, as they did for the Swamp Gulch wetland cores, and there was a subsequent decline in sulfate concentrations in the effluent, suggesting that leaching of excess sulfate had occurred in these cores. These high levels of sulfate suggest that sulfate reduction was not taking place in the Sand Coulee wetland soils. This could be due to the absence of sufficient carbon to allow the sulfate-reducing bacteria to proliferate (Lovley 1987). It could also be due to the acidity of the cores as sulfate-reducing bacteria do not grow well at pH levels < six (Widdel and Pfennig 1984).

In the SC-CH core, chloroform may have killed the sulfate-reducing bacterial population that was present. Alternatively, chloroform treatment may have enriched other microorganisms, effectively displacing sulfate-reducing bacteria. Since the other Sand Coulee wetland cores did not reflect lower sulfate levels, it may have been that sulfate-reduction was not occurring as a consequence of unknown factors.

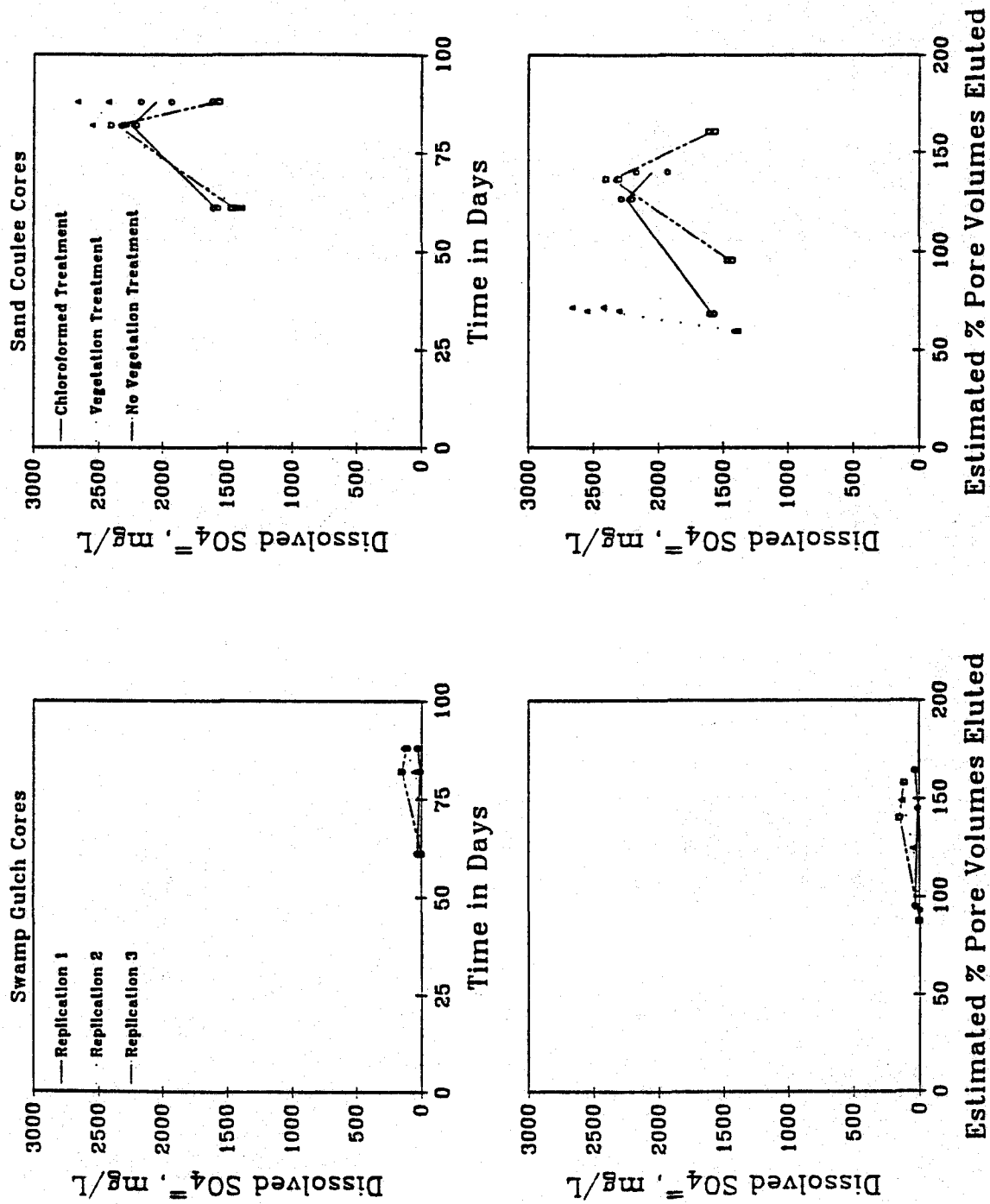


Figure 10-9. Swamp Gulch and Sand Coulee soil core effluent (eluate) sulfate concentrations measured over study period.

Table 10-2. Swamp Gulch sulfate concentrations vs. flow rate.

Core Number	Flow rate ¹ (mL/day)	Mean sulfate ² (mg/L)
SG-1	219.3	35.3 ± 3.0
	401.7	12.7 ± 3.6
	472.7	33.9 ± 2.5
SG-2	213.5	0.0 ± 0.0
	255.7	49.0 ± 6.2
	596.4	130.5 ± 9.2
SG-3	198.7	6.2 ± 5.7
	424.3	152.5 ± 7.8
	441.4	115.5 ± 2.1

¹ n = 1
² n = 2

Table 10-3. Sand Coulee sulfate concentrations vs. flow rate.

Core Number	Flow rate ¹ (mL/day)	Mean sulfate ² (mg/L)
SC-V	144.2	1399.0 ± 19.8
	82.4	2425.0 ± 173.9
	91.4	2540.0 ± 169.7
SC-VN	243.9	1450.5 ± 26.2
	330.4	2353.5 ± 72.8
	584.3	1585.0 ± 35.4
SC-CH	172.0	1588.0 ± 25.5
	468.3	2245.0 ± 59.4
	340.0	2050.0 ± 169.7

¹ n = 1
² n = 2

These findings indicate that sulfate remediation at the Sand Coulee constructed wetland was not occurring. In order to optimize wetland performance, construction specifications should favor the proliferation and activities of sulfate-reducing bacteria. This means that:

- Anaerobic conditions must prevail within the sediment. Therefore slow flow rates and microbial population diversity enhancement is important. Varying wetland depth would also be beneficial.
- Metabolizable carbon and other nutrients should be enhanced. This would include providing living plants and fertilizer. This would also increase the concentrations of aerobic microorganisms (increased species diversity) whose activities would lead to more favorable conditions for the anaerobic sulfate-reducing bacteria (by removing oxygen and providing alternate carbon substrates).
- For the sulfate-reducing bacteria to proliferate, the pH of the constructed wetland should be near six or greater. Therefore it would be prudent to select wetland construction materials that have an initial pH near six or above. The use of lime in a constructed wetland can help attain this goal.

10.3.4.3 Iron determinations

Iron analyses show a consistent total dissolved iron concentration in the influent AMD of 10.2 to 16.9 mg/L, averaging 13.3 mg/L (Figure 10-8). This iron was present in the ferric form, which was not surprising, since the source of the acid mine drainage was a shallow pond.

The presence of ferrous iron in the effluent of the soil cores was not surprising, since the soil cores provide a reducing environment. It was unexpected that no apparent iron remediation occurred in any of the cores since all six of the wetland cores eluted more iron than was added to them by the eluent AMD. Conceivably the organic material acts as a cation exchange column. In the case of the Swamp Gulch wetland soils there appear to be hydrogen ions displacing metal ions at the exchange sites. These three cores remained fairly stable in their effluent iron concentrations over time. The eluate iron concentrations ranged from 77 to 196 mg/L for the Swamp Gulch wetland cores. These concentrations did not decrease over time or over % pore volume eluted (Figure 10-10), nor did flow rate affect them.

In the Sand Coulee wetland cores the iron concentrations averaged 203 ± 136.6 mg/L. The iron concentrations of the SC-VN

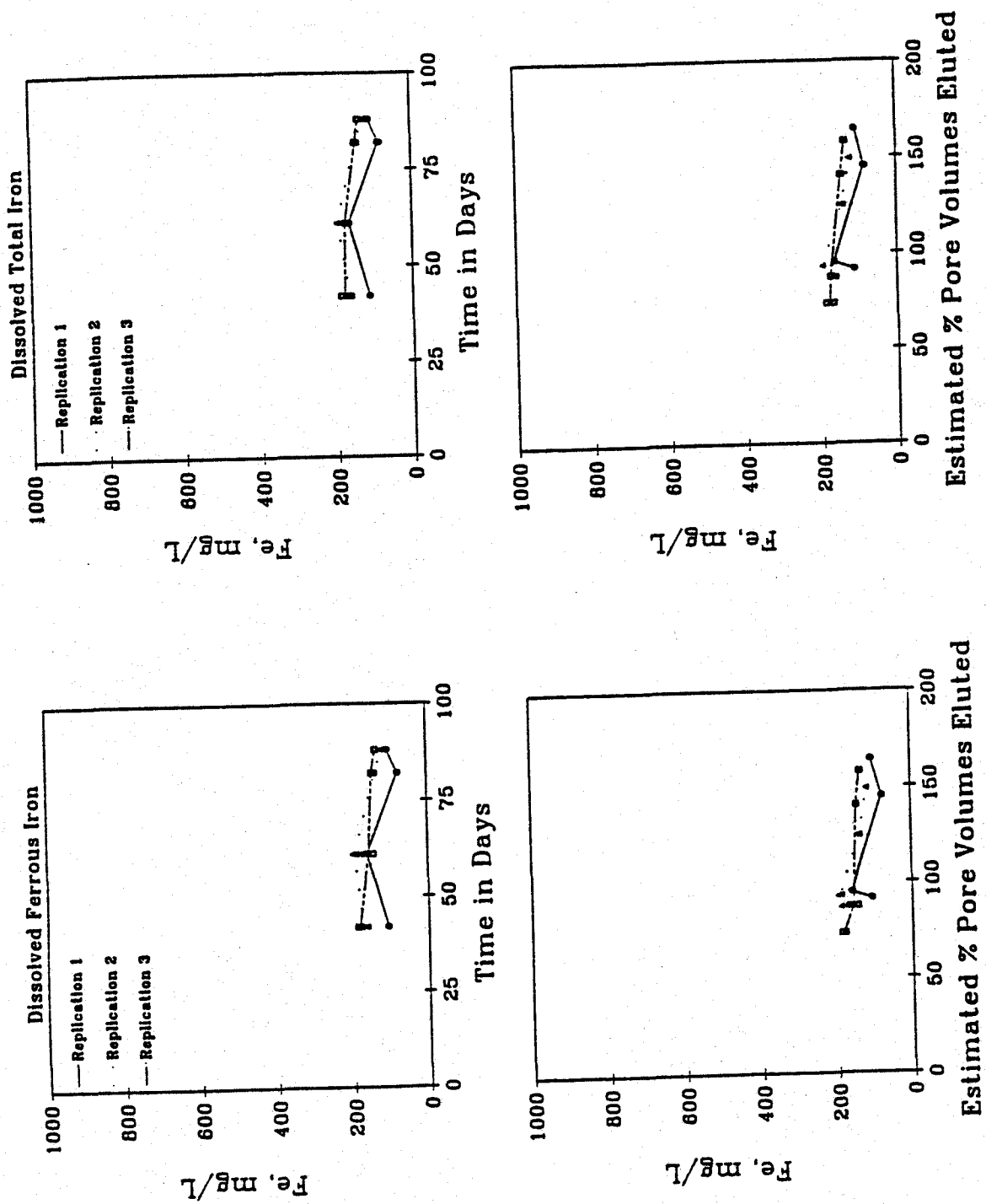


Figure 10-10. Swamp Gulch soil core effluent (eluate) dissolved total and ferrous iron concentrations measured over study period.

(no vegetation) core effluent constantly declined (Figure 10-11). On day 42, the concentration of iron in this core effluent was 476 mg/L. By day 88, it had decreased to 35.5 mg/L. The SC-V (vegetation) and SC-CH (chloroform treated) cores were similar except that their iron concentrations were > 100 mg/L at the end of the study. The SC-CH core had high iron concentrations initially (450 mg/L) and began to decrease at a later date than the other cores (day 62). By the final iron analysis, the Sand Coulee wetland iron levels had dropped by half.

Thus all of the Swamp Gulch and Sand Coulee soil core eluate iron concentrations were higher than those of the influent (eluent) AMD. However, cores from each of the sites exhibit different trends. The Swamp Gulch wetland cores released about the same concentration of iron over time, irrespective of: 1) flow rate or, 2) percent pore volumes eluted. The Sand Coulee wetland cores eluted extremely high concentrations of iron early in the study, with a tendency toward decreasing iron concentrations with time. This may be related to flow rate or it may be a washing out of iron present prior to the beginning of the study.

We suggest different hypotheses for these phenomena. In the Swamp Gulch wetland cores no apparent iron remediation took place. The iron seems to be released from the soil at a fairly steady rate suggesting that cation exchange with acidity (hydrogen ions) is occurring.

In the case of the Sand Coulee wetland cores, the iron concentrations decreased over time. This could be indicative of either, 1) leaching of iron by the Swamp Gulch AMD, 2) or these cores may be remediating iron. Of these suppositions, it seems most likely that iron is being leached. When the total effluent volumes for the SC-V and SC-VN cores are compared with one another (Table 10-4), the iron concentration in the effluent decreases as the flow through volume increases. In the SC-CH core, the chloroform may have altered the iron remediation properties.

These findings indicate that iron removal at the natural wetland and at the constructed wetland sites was not occurring. In order to promote iron removal in constructed wetlands, materials should be selected that:

- Provide as much cation exchange capacity as possible to remove acidity and metals.
- Provide favorable conditions for the proliferation of sulfate-reducing bacteria. If these bacteria are able to reduce sulfate to sulfide, metals will be immobilized through to formation of metal sulfide precipitates.

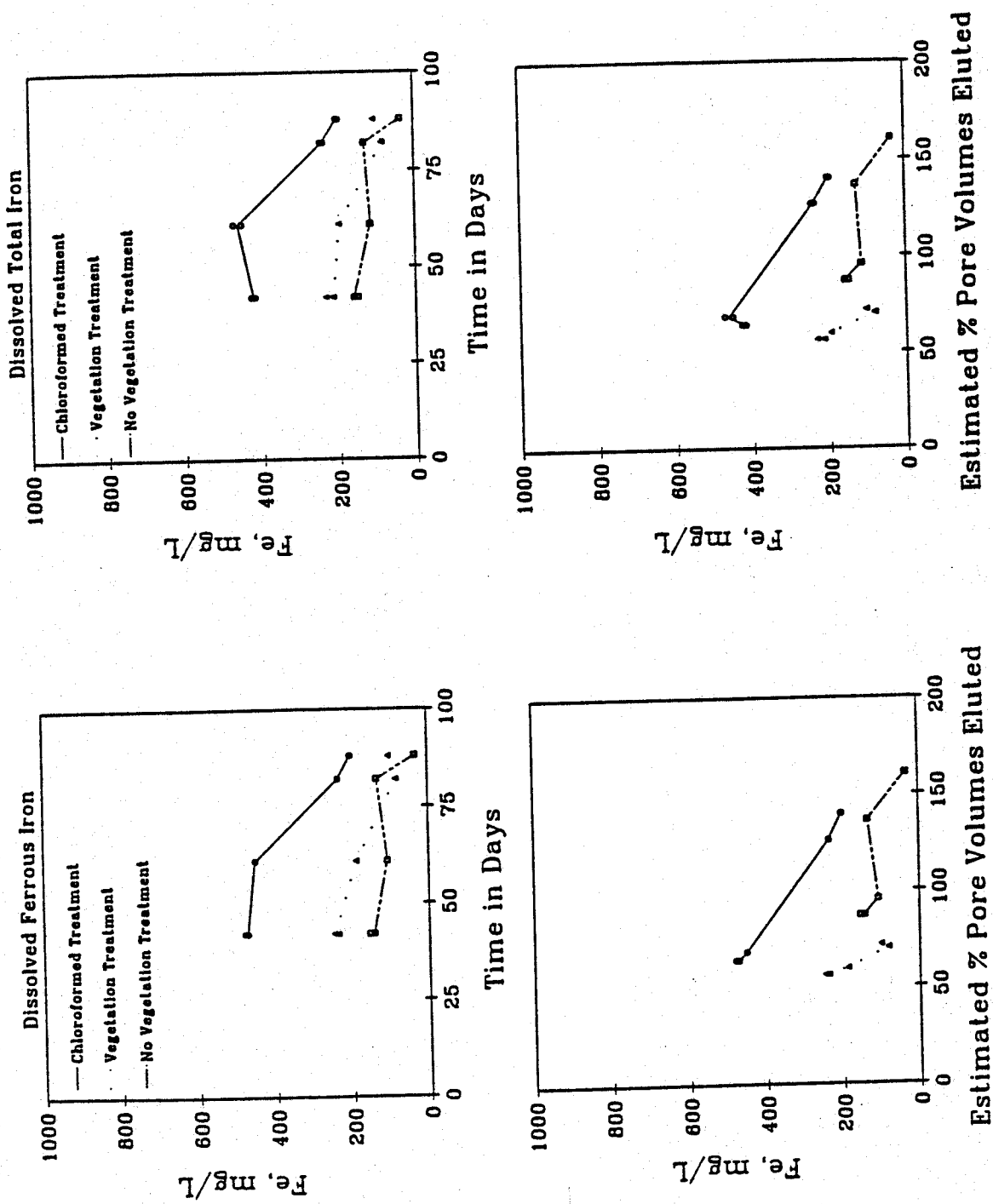


Figure 10-11. Sand Coulee soil core effluent (eluate) dissolved total and ferrous iron concentrations measured over study period.

Table 10-4. Sand Coulee iron concentrations vs. total flow volume eluted at end of study.

<u>Core Number</u>	<u>Volume AMD¹ Eluted, mL</u>	<u>Average Iron² mg/L</u>
SC-V	12,453	104.2 ± 1.1
SC-VN	27,180	35.5 ± 0.0
SC-CH	23,768	199.2 ± 3.5

¹ n = 1

² n = 2

10.4 Phase III: MICROBIAL BIOMASS, COMMUNITY STRUCTURE AND PHYSIOLOGICAL STATUS ASSESSMENT OF NATURAL AND CONSTRUCTED WETLANDS

10.4.1 Introduction

Quantitative extraction of cellular components of sedimentary microbiota permits an assessment of the viable microbial community biomass, composition and, to a degree, physiological status, without the problems associated with direct enumeration or cultural methods (White 1986). In particular, phospholipid ester-linked fatty acids (PLFA) have proven to be reproducible, capable indicators of a wide suite of microbiota encountered in the environment (e.g., Guckert et al. 1985). The phosphate of the microbial phospholipids has been shown to have a rapid turnover in both living and killed cells (White et al. 1979), indicating that an accurate analysis selective for this lipid class would permit a quantitative estimation of the viable microbiota. The objective of these studies was to use PLFA analysis to assist in the assessment of the role of microorganisms in the amelioration of AMD in natural and constructed wetlands by demonstrating the range of biomasses and the differences in the structure and physiological status of the viable microbial communities within these wetlands.

10.4.2 Technical Approach

A total of twenty wetland sediment samples were collected in the field and subjected to PLFA analysis in conjunction with the wetlands study.

10.4.3 Materials and Methods

10.4.3.1 Field sampling

Sediment samples were collected from several wetland sites: Hardscrabble Creek, Swamp Gulch and Sand Coulee (Figure 10-1). Hardscrabble Creek is a natural wetland site. The Swamp Gulch wetland site receives inputs of AMD (pH = 2.5) from the Carbonate mine. Hardscrabble Creek was chosen as a "control site" that did not receive AMD, but was in close proximity to Swamp Gulch. Sediment samples were collected from the 20-21 cm depth horizon for both sites in individual replicate cores. The Sand Coulee wetlands are constructed wetland cells which began operation within the past two years. In both Sand Coulee wetland cells studied, the surface sediment (approximately 1-2 cm) and 20-21 cm depth horizon were sampled in independent cores collected along a transect across the wetlands.

The principal differences in the wetland sediments observed included the dense vegetation mat and complex root systems of the natural wetlands which held the wetland sediment together into a cohesive unit, whereas the man-made wetland sediment lacked this vegetation and cohesiveness. All wetland sediment samples were frozen and then lyophilized (freeze-dried). After drying, the samples were inventoried, randomly relabeled, and total dry weights were determined.

10.4.3.2 Statistical analyses

All statistical analyses utilized SPSSX statistics programs available on the Montana State University Honeywell 66/DPS mainframe computer with CP-6 operating system. Data transformations were used to meet the homogeneity of error variance assumptions for the analysis of variance model used (Winer 1971). In general, a \log_{10} transformation was performed, except when the variable was a percentage, in which case an arcsine square root transformation was made (Winer 1971). Following the analysis of variance (ANOVA), Tukey's Honestly Significant Difference test (HSD, SPSSX programs) was calculated for a multiple comparison of means. Significant difference maps were produced keeping the within-experiment, family-wise error rate set at $\alpha = 0.05$.

10.4.3.3 Lipid nomenclature

Fatty acids are named as the total number of carbon atoms, followed by the number of double bonds closest to the aliphatic (w) end with the two numbers being delimited with a colon (i.e., carbon atom number:double bond number). Furthermore, the geometry ('c' for cis and 't' for trans) of the double bond is also denoted with the double bond number (e.g., 18:1w7c). The prefixes 'i', 'a', and 'br' refer to iso, anteiso and methyl-

branching of the unconfirmed position, respectively. Cyclopropyl fatty acids are designated as 'cy' with the ring position in parenthesis relative to the aliphatic end (e.g., cy 19:0).

10.4.4 Results

10.4.4.1 Total Biomass

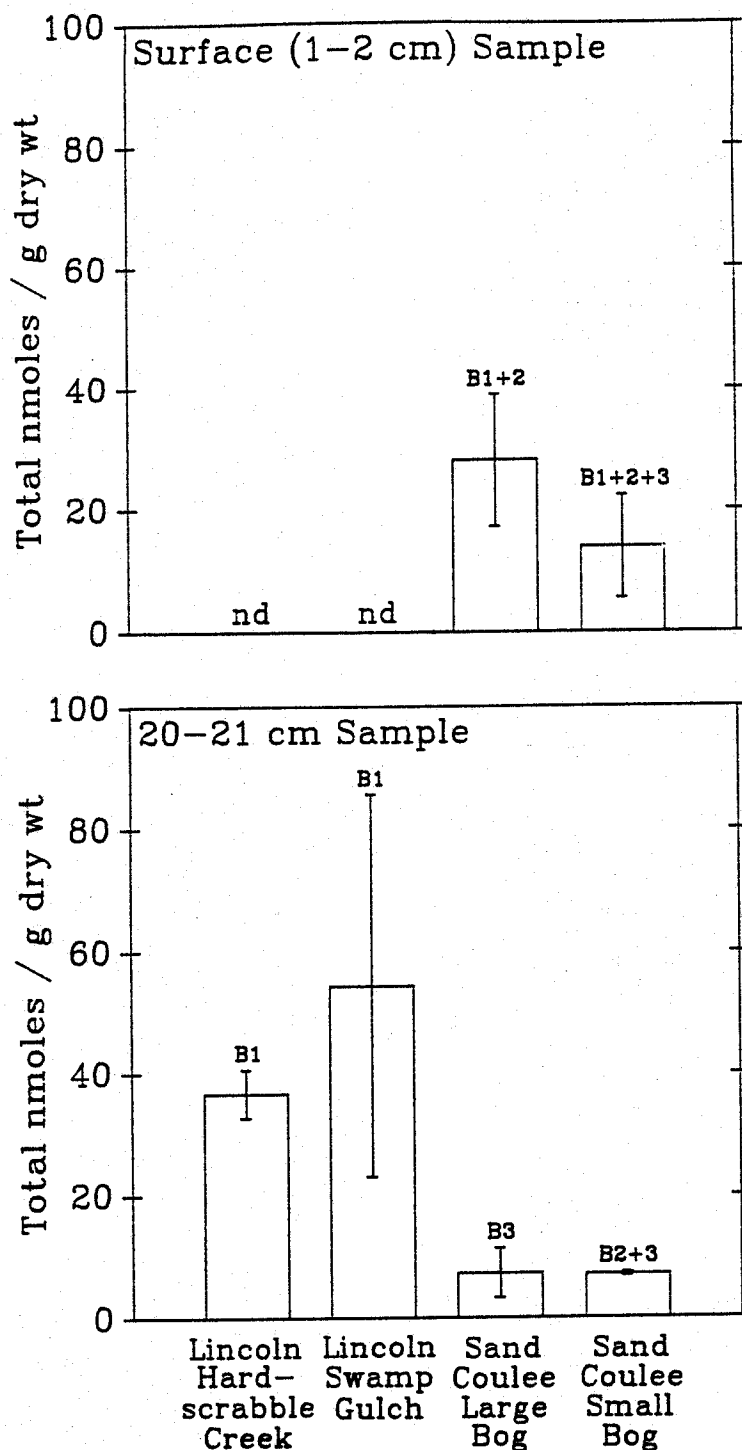
The total PLFA quantified can be used as an estimate of the biomass of the viable microbiota in the wetland sediments. The total nmoles of PLFA per gram dry weight of wetland sediment is shown for each site and depth horizon in Figure 10-12. Histograms which are labeled with symbols for the same group (e.g., B1 and B1+2, Figure 10-12) are included in the same homogeneous subset as defined by the Tukey HSD test. Histograms with no common symbols (e.g., N1 and N2, Figure 10-13) are significantly different by the Tukey test. The letters for these symbols indicate the parameter measured (B=biomass, N=nmoles/gdw, M=mole percent, T=trans/cis, C=cyclopropyl/cis). The numbers differentiate the groups, with 1 being the group with the highest mean value of the parameter measured.

At the 20 cm depth horizon, therefore, the Hardscrabble Creek and Swamp Gulch sites have a significantly greater microbial biomass than the Sand Coulee sites (Figure 10-12). The 20 cm horizons for the Lincoln sites are as dense in microbial biomass as the surface sediments of the Sand Coulee sites.

10.4.4.2 Community structure

In addition to estimating total microbial biomass, the PLFA profiles can be used to compare the microbial community structure for the sampled wetlands. Table 10-5 includes the PLFA profiles for all sites and depth horizons expressed as the mole percent of the entire PLFA profile. The fatty acids are listed in their elution order off the chromatography column. The data are expressed as average \pm one sample standard deviation. Statistically significant differences within individual PLFA are marked with an '*' (ANOVA, $\alpha = 0.05$).

While the presentation of a complete data set such as shown in Table 10-5 is important, microbial community structure interpretations are difficult using individual PLFA. Table 10-6 describes some PLFA groups assembled to aid interpretation. The eukaryotes are characterized by polyunsaturated fatty acids (Shaw 1966). The bacterial group is a combination of the terminally-branched fatty acids common to gram-positive-like organisms (Kaneda 1977); the PLFA indicative of sulfate-reducing bacteria (Dowling et al. 1986); the cyclopropyl fatty acids and their immediate precursor, the major product of the anaerobic desaturase fatty acid biosynthetic pathway, 18:1w7c (Fulco

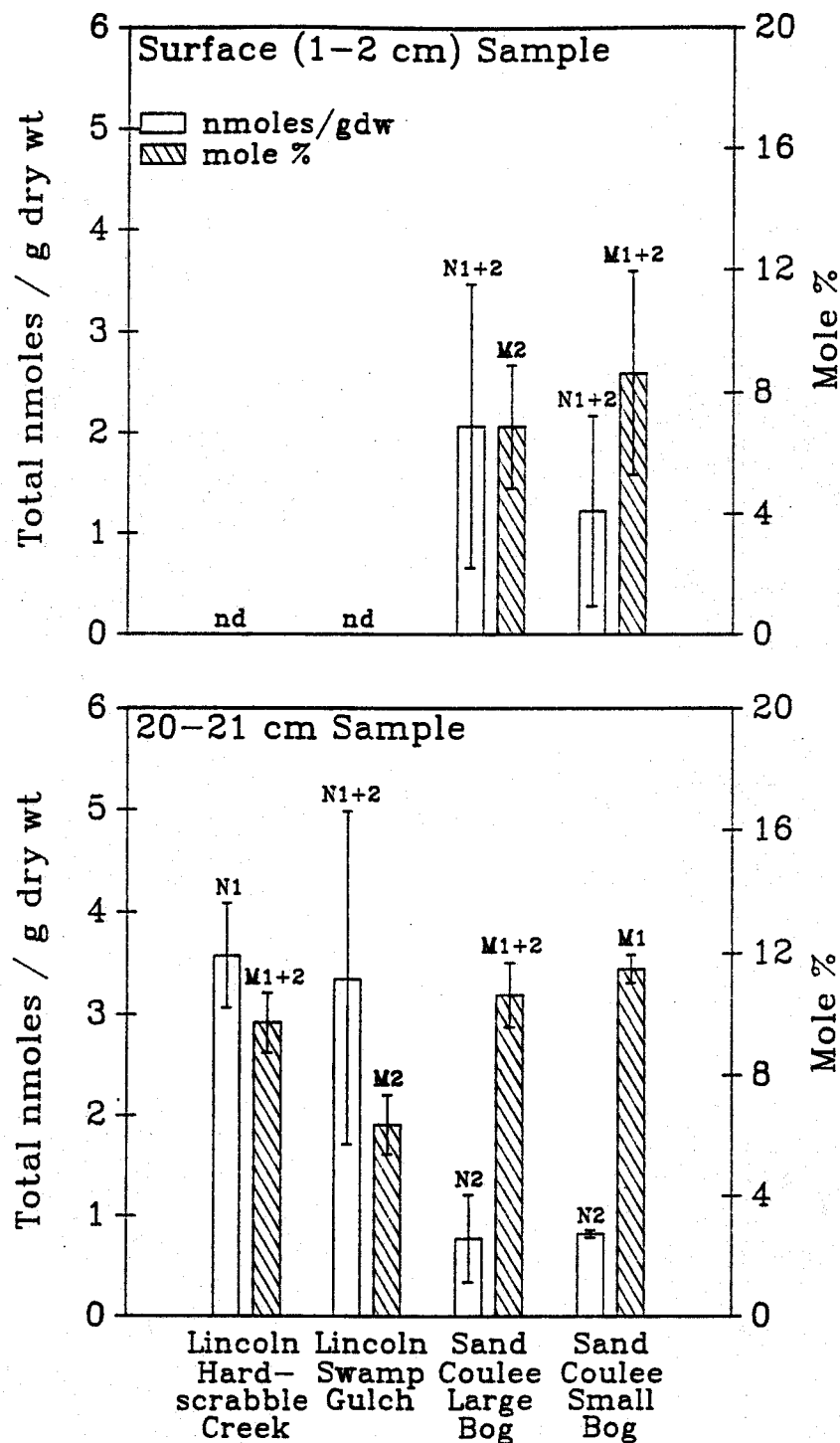


nd = not determined.

Histograms represent means \pm 1 std. deviation.

B1+2+3 symbols used to distinguish subsets generated by Tukey's test of \log_{10} transformed data (see text for details).

Figure 10-12. Total microbial biomass as measured by analysis of phospholipid ester-linked fatty acids.



nd = not determined.

Histograms represent means \pm 1 std. deviation.

N1+2 symbols used to distinguish subsets generated by Tukey's test of \log_{10} transformed data; M1+2 used $2 \cdot \arcsin$ square root transformed data (see text for details).

Figure 10-13. Sulfate-reducing bacterial phospholipid fatty acid biomarker expressed as density per gram of soil and proportion of total microbial biomass.

Table 10-5. Fatty Acid (FA) mole % for the various sampling sites.

FA	a=0.05	LINCOLN				SAND COULEE LARGE CELL		SAND COULEE SMALL CELL	
		Hardscrabble	Swamp	Surface	20cm	Surface	20cm	Surface	20cm
i14:0	*	0.6 + 0.0	1.5 + 0.4	0.2 + 0.1	0.8 + 0.5	0.1 + 0.2	0.5 + 0.1	0.1 + 0.2	0.5 + 0.1
i14:0	*	0.7 + 0.0	1.3 + 0.2	5.7 + 2.4	3.6 + 1.2	6.8 + 4.6	1.4 + 0.2	1.4 + 0.2	1.4 + 0.2
i15:0	*	2.0 + 0.0	3.7 + 0.0	2.4 + 0.0	5.3 + 1.9	2.9 + 0.7	4.7 + 0.8	4.7 + 0.8	4.7 + 0.8
a15:0		3.9 + 0.2	6.2 + 0.6	2.3 + 0.6	3.3 + 2.6	2.0 + 0.4	4.3 + 0.6	4.3 + 0.6	4.3 + 0.6
i15:1		0.2 + 0.0	0.2 + 0.2	0.2 + 0.3	0.7 + 0.9	0.1 + 0.1	0.1 + 0.1	0.1 + 0.1	0.1 + 0.1
i15:0		0.6 + 0.0	0.6 + 0.0	3.6 + 3.8	0.9 + 0.7	1.9 + 0.7	0.5 + 0.0	0.5 + 0.0	0.5 + 0.0
i16:0		0.8 + 0.6	1.3 + 0.3	1.7 + 0.1	1.7 + 0.8	1.0 + 0.1	1.4 + 0.2	1.4 + 0.2	1.4 + 0.2
i16:1w9c	*	1.4 + 0.2	0.8 + 0.3	0.7 + 0.4	1.3 + 0.4	0.9 + 0.2	1.5 + 0.3	1.5 + 0.3	1.5 + 0.3
i16:1w7c		8.6 + 0.4	6.8 + 2.0	9.5 + 4.8	4.7 + 1.4	7.5 + 2.0	5.8 + 0.2	5.8 + 0.2	5.8 + 0.2
i16:1w7t		1.1 + 0.0	2.4 + 0.8	0.2 + 0.2	0.2 + 0.2	0.0 + 0.0	0.6 + 0.0	0.6 + 0.0	0.6 + 0.0
i16:1w5c	*	2.4 + 0.1	1.5 + 0.3	0.8 + 0.5	1.3 + 0.6	0.6 + 0.2	1.3 + 0.4	1.3 + 0.4	1.3 + 0.4
i16:1w13t	*	0.0 + 0.0	0.2 + 0.2	0.1 + 0.1	1.7 + 1.7	0.4 + 0.4	0.4 + 0.6	0.4 + 0.6	0.4 + 0.6
i16:0	*	10.6 + 0.3	12.1 + 0.4	20.4 + 2.0	13.4 + 1.3	13.7 + 1.6	11.5 + 1.1	11.5 + 1.1	11.5 + 1.1
i17:1w7c	*	1.6 + 0.1	1.2 + 0.2	1.3 + 0.3	2.4 + 0.7	1.4 + 0.4	1.8 + 0.4	1.8 + 0.4	1.8 + 0.4
i10mel6:0	*	6.5 + 0.7	2.4 + 0.3	3.8 + 1.9	3.7 + 0.5	1.9 + 0.2	2.8 + 0.6	2.8 + 0.6	2.8 + 0.6
i17:0	*	0.8 + 0.0	0.8 + 0.1	1.2 + 0.5	1.8 + 0.3	1.1 + 0.1	1.4 + 0.3	1.4 + 0.3	1.4 + 0.3
a17:0/17:1w9c		2.1 + 0.2	1.6 + 0.0	2.9 + 1.5	2.3 + 0.4	1.8 + 0.4	1.7 + 0.1	1.7 + 0.1	1.7 + 0.1
i17:1w6c		1.2 + 0.2	0.2 + 0.2	0.2 + 0.2	0.5 + 0.8	0.4 + 0.6	0.1 + 0.2	0.1 + 0.2	0.1 + 0.2
cy17:0	*	1.7 + 0.0	2.8 + 1.0	1.8 + 0.5	4.8 + 0.2	5.4 + 3.3	7.1 + 0.5	7.1 + 0.5	7.1 + 0.5
i17:0		0.7 + 0.0	0.6 + 0.1	0.6 + 0.4	0.7 + 0.2	0.6 + 0.2	0.6 + 0.2	0.6 + 0.2	0.6 + 0.2
i18:3w6		1.1 + 1.0	1.1 + 0.7	1.4 + 0.7	0.7 + 0.5	1.3 + 0.3	0.4 + 0.1	0.4 + 0.1	0.4 + 0.1
i18:2w6	*	5.5 + 0.9	5.7 + 0.1	4.1 + 1.9	1.8 + 0.2	6.7 + 2.8	1.7 + 0.2	1.7 + 0.2	1.7 + 0.2
i18:3w3		0.5 + 0.8	0.6 + 0.6	0.3 + 0.7	0.2 + 0.3	0.8 + 1.4	0.0 + 0.0	0.0 + 0.0	0.0 + 0.0
i18:1w9c	*	7.8 + 0.1	7.8 + 1.1	5.4 + 0.6	6.0 + 1.3	7.5 + 1.7	7.2 + 1.2	7.2 + 1.2	7.2 + 1.2
i18:1w7c	*	25.8 + 0.4	18.9 + 2.0	12.3 + 2.6	14.0 + 2.0	13.6 + 1.9	16.8 + 1.1	16.8 + 1.1	16.8 + 1.1
i18:1w7t	*	0.6 + 1.0	3.9 + 1.1	0.3 + 0.3	0.7 + 0.5	0.6 + 0.1	1.1 + 0.1	1.1 + 0.1	1.1 + 0.1
i18:1w5c	*	0.1 + 0.2	0.3 + 0.2	0.2 + 0.2	1.0 + 1.1	0.4 + 0.4	0.2 + 0.4	0.2 + 0.4	0.2 + 0.4
i18:0	*	2.1 + 0.1	2.2 + 0.2	1.8 + 0.0	3.5 + 0.6	2.7 + 1.0	3.2 + 0.2	3.2 + 0.2	3.2 + 0.2
br19:1	*	1.2 + 0.2	0.5 + 0.4	0.5 + 0.2	1.6 + 0.2	1.0 + 0.4	2.1 + 0.4	2.1 + 0.4	2.1 + 0.4
cy19:0	*	8.3 + 1.2	10.9 + 6.6	7.4 + 6.3	15.9 + 3.7	10.7 + 2.2	19.3 + 2.7	19.3 + 2.7	19.3 + 2.7
20:4w6	*	0.4 + 0.0	0.1 + 0.1	3.1 + 0.8	0.6 + 0.5	2.9 + 1.6	0.4 + 0.1	0.4 + 0.1	0.4 + 0.1
20:5w3	*	0.2 + 0.1	0 + 0.1	4.0 + 3.8	0.4 + 0.4	2.2 + 1.0	0.1 + 0.2	0.1 + 0.2	0.1 + 0.2

1983). The anaerobic desaturase group includes the w7c products and the corresponding cyclopropyl and mid-chain branched isomers resulting from transmethylation, as well as the trans-isomers which are hypothesized to be formed by an isomerase of the w7c products of anaerobic desaturase (Guckert et al. 1986). The sulfate-reducing bacteria can also be further defined based on the genera-specific PLFA biomarkers (Dowling et al. 1986; Edlund et al. 1985).

When the PLFA groups defined in Table 10-6 are used to test for changes in microbial community structure, the significant difference map shown in Table 10-7 is produced. The map is set up such that treatment means for each PLFA group increase from left to right, and those connected by a common line segment are not significantly different by the Tukey test ($\alpha = 0.05$).

The results presented in Table 10-7 will be discussed in more detail later. Briefly, the Sand Coulee wetland surface depth horizons (site codes = 3 and 5) were characterized as having the highest eukaryotic and lowest bacterial proportions. There were no significant differences in the proportions of gram-positive-like organisms or the total anaerobic desaturase

Table 10-6. Description of phospholipid ester-linked fatty acids (PLFA) used to define microbial groups for data interpretation.

MICROBIAL GROUP	PLFA USED TO DEFINE GROUP
Eukaryotes	20:4w6 + 20:5w3 + 18:3w3 + 18:2w6 + 18:3w6 + 16:1w13t
Bacteria	i15:0 + a15:0 + i17:1w7c + 10me16:0 + i17:0 + a17:0/17:1w9c + cy17:0 + 18:1w7c + cy19:0
Gram-positive	i15:0 + a15:0 + i17:0 + a17:0/17:1w9c
Anaerobic Desaturase	16:1w7c + 16:1w7t + 10me16:0 + cy17:0 + 18:1w7c + 18:1w7t + cy19:0
18:1w7c	18:1w7c
Sulfate Reducing Bacteria	i17:1w7c + 10me16:0 + cy17:0
<u>Desulfobacter</u> spp.	10me16:0 + cy17:0
<u>Desulfovibrio</u> spp.	i17:1w7c

products. However, the principal unmodified product of anaerobic desaturase, 18:1w7c, was in the greatest proportion in the Hardscrabble Creek site (site code = 1). All sulfate-reducing bacterial groups were in the greatest proportion of the total microbial biomass in the Sand Coulee 20 cm depth horizons (site codes 4, 6) with the lowest proportions being in the Swamp Gulch 20 cm (site code 2) and Sand Coulee surface (site codes 3, 5).

Since sulfate-reducing bacterial biomass and activity had been assumed to have a possible influence on AMD amelioration, a further illustration of sulfate-reducing bacterial distribution between the sites was done (Figure 10-13). The mole percent histograms graphically display the relationship shown in the significant difference map (Table 10-7). The total sulfate-reducing bacterial biomass estimates in nmole per gram dry weight wetland sediment are also shown indicating that although the Swamp Gulch site has the lowest proportion of biomass as sulfate-reducing bacteria (group M2, Figure 10-13), the density of sulfate-reducing bacteria (per gram dry weight) for this site along with Hardscrabble Creek is included in the highest group (group N1, Figure 10-13). In general, the Sand Coulee sites are lower in sulfate-reducing bacterial biomass density with the higher values occurring in the surface sediments. Greater total biomass and species diversity was evident at the Hardscrabble Creek and Swamp Gulch wetland sites.

Even though sulfate-reducing bacteria are present in the Sand Coulee constructed wetland they are ineffective in removing sulfate. Where these organisms are present (at the surface) environmental conditions (e.g., O₂ levels, nutrient availability) would preclude their activity. Therefore, specifications for wetland construction should address carbon and other nutrient additions as well as reduced oxygen levels necessary to enhance the densities and activities of these organisms.

10.4.4.3 Physiological stress

In addition to microbial biomass and community structure estimates the PLFA can be used to describe aspects of the community physiological status. Specifically, the trans/cis ratio of monoenoic PLFA and the cyclopropyl/cis PLFA ratios have been suggested as possible "starvation or stress" lipid indices (Guckert et al. 1986). These ratios for each wetland site and depth horizon are shown in Figure 10-14. The Sand Coulee sites can all be characterized by having higher cyclopropyl/cis ratios (group C1, Figure 10-14), whereas the Swamp Gulch site is characterized by an extremely high trans/cis ratio (group T1, Figure 10-14). The control Hardscrabble Creek site is low for both of these "stress" lipid ratios (groups T2 and C2, Figure 10-14).

Table 10-7. Significant difference map generated from Tukey's HSD test¹.

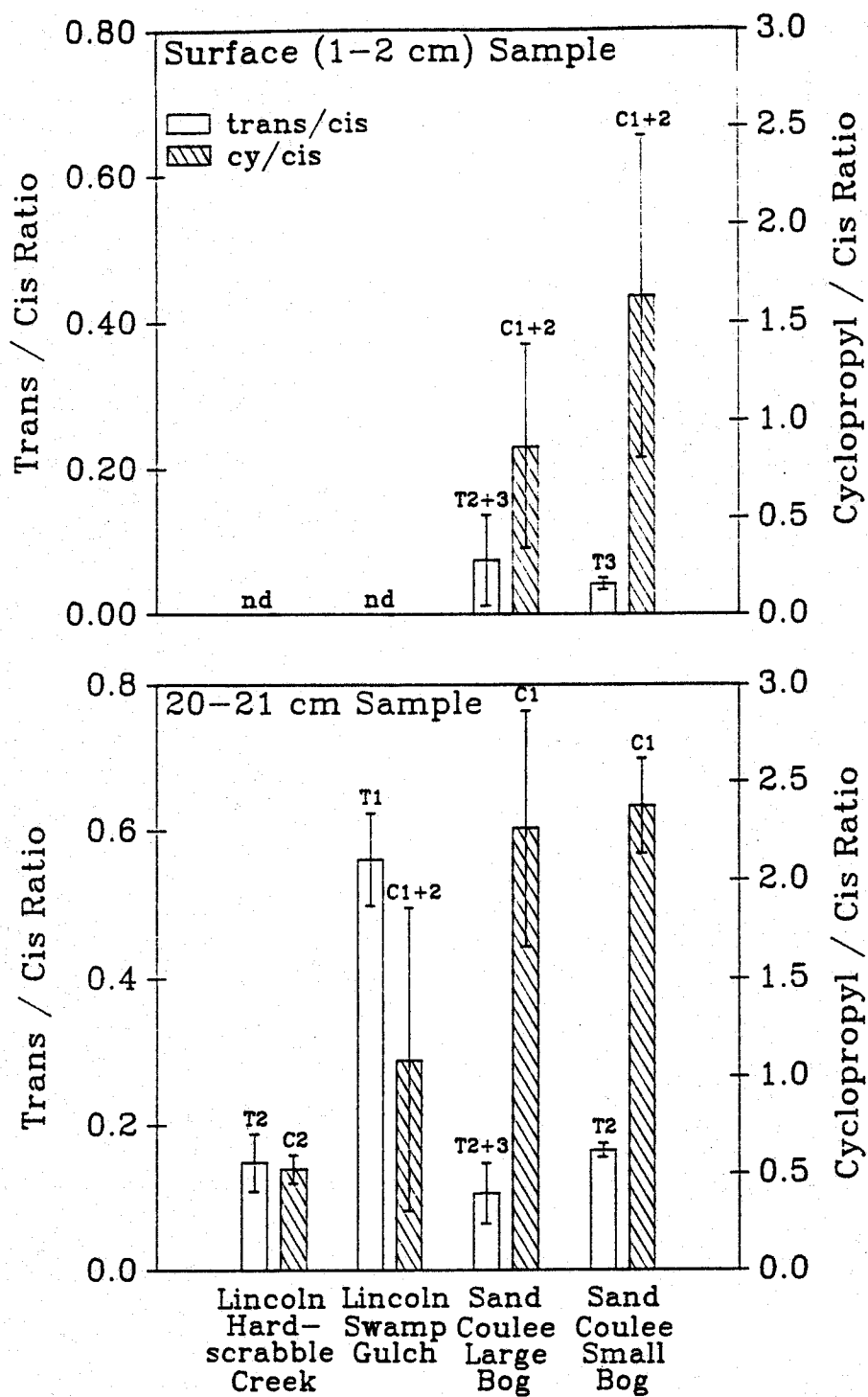
Site
Codes:

- 1 = Lincoln, Hardscrabble Creek (20 cm)
- 2 = Lincoln, Swamp Gulch (20 cm)
- 3 = Sand Coulee Large Bog (surface)
- 4 = Sand Coulee Large Bog (20 cm)
- 5 = Sand Coulee Small Bog (surface)
- 6 = Sand Coulee Small Bog (20 cm)

MICROBIAL GROUP ²	Low High					
Eukaryotes	6	4	1	2	3	5
Bacteria	3	5	2	4	1	6
Gram Positive (branched chains)	5	3	1	6	2	4
(Anaerobic desaturase)	3	5	4	2	1	6
18:1w7c	3	5	4	6	2	1
Sulfate Reducing Bacteria	2	3	5	1	4	6
<i>Desulfobacter spp.</i>	2	3	5	1	4	6
<i>Desulfovibrio spp.</i>	2	3	5	1	6	4

¹ Within experiment familywise error rate set at $\alpha=0.05$ for 2*arcsine square root transformed mole % data of Table 10.5 grouped according to designations in Table 10.6.

² Treatment means for each group of fatty acids increase from left to right, and those connected by a common line segment are not significantly different for this test.



nd = not determined.

Histograms represent means \pm 1 std. deviation.

T1+2+3 and C1+2 symbols used to distinguish subsets generated by Tukey's test of \log_{10} transformed data (see text for details).

Figure 10-14. Microbial community trans/cis and cyclopropyl/cis phospholipid ester-linked fatty acids ratios.

10.4.5 Discussion

The results of the PLFA data presented here need to be discussed in the context of results of the other projects conducted within the wetlands study. The Swamp Gulch wetland site has had in situ AMD inputs. The natural wetland sediments (Phase III), as well as sediment cores brought into the laboratory and continually flushed with AMD (Phase II), are able to effectively buffer the AMD water. The cores analyzed in the laboratory have been shown to have a concomitant removal of sulfate from the AMD during the buffering process. When sediment cores from the Sand Coulee large wetland cell were exposed to AMD, no buffering occurred except when a core was treated with chloroform. No apparent sulfate utilization was measured from the Sand Coulee wetland cores.

All of these wetland sediments are bacterially dominated communities (Table 10-5, note proportions of bacteria PLFA as described in Table 10-6) as opposed to a prevalence of eukaryotic (i.e., higher life forms such as green algae, fungi, etc.) organisms. The Sand Coulee wetland surface horizons show a higher eukaryotic input (Table 10-7). The natural wetlands, which contain more vegetation and root material, support a much higher microbial biomass density than the constructed wetlands of the Sand Coulee cells (Figure 10-12).

The apparent correlation of sulfate loss with AMD buffering suggests an important role for sulfate-reducing bacteria in the AMD amelioration process. Sulfate-reducing bacteria data displayed in Figure 10-13 indicate that, as was seen for total microbial biomass, the natural wetlands (Hardscrabble Creek, Swamp Gulch) support a higher sulfate-reducing bacterial biomass density than the artificial wetlands. Interestingly, the sulfate-reducing bacterial proportion of the total microbial biomass shows the opposite trend with the non-buffering Sand Coulee site having a higher proportion of the total microbial community as sulfate-reducing bacteria. The data in Figure 10-13 suggest several important aspects of AMD buffering by wetland systems: 1) If total sulfate-reducing bacterial biomass density is an important factor in the buffering process, we might predict that the Hardscrabble Creek control site would also be able to adequately buffer AMD; 2) The higher proportion of sulfate-reducing bacteria in the less diverse Sand Coulee microbial communities with the lower sulfate-reducing bacterial densities suggests the importance of other microbial populations to the sulfate-reducing bacteria. Sulfate-reducing bacteria are obligate anaerobes which are considered to be terminal carbon users when sufficient sulfate levels are present. Organisms at this nutritional level rely upon other aerobic, facultatively anaerobic and obligately anaerobic primary degraders to generate the carbon and energy substrates required for sulfate-reducing bacterial growth from the more complex molecular inputs into the

wetland (Dowling 1987). Therefore, in the constructed wetland the availability of additional simple carbon sources (e.g., viable plants) would be beneficial to the sulfate-reducing bacterial populations.

This discussion suggests that if the primary degraders could be stimulated by an input of carbon and energy sources the sulfate-reducing bacterial populations might be better "fed," thereby increasing their density. If the density of sulfate-reducing bacterial populations is actually associated with AMD buffering, this artificial wetland system might then provide acidity and sulfate amelioration. One obvious difference in the wetlands that might account for the differences in microbial population proportions is the lack of vegetation in the Sand Coulee wetlands. Root systems provide nutritional inputs. Also, bioturbation of burrowing micro-eukaryotes associated with this vegetation can have an effect on the wetland sediment. Note that after the Sand Coulee surface horizons the Swamp Gulch and Hardscrabble Creek sites also have higher proportions of eukaryotic biomass than the corresponding 20 cm depth horizons at the Sand Coulee cells (Table 10-7).

It should not be assumed that wetland size alone governs performance. These findings indicate that the amount of microbial biomass is deficient in constructed wetlands when compared to natural wetlands. Enhancement of the microbial components would be expected to markedly affect the amelioration of sulfate and acidity due to AMD.

The PLFA group made up of the products of the anaerobic desaturase fatty acid biosynthetic pathway shows no significant differences in their proportion to the overall microbial community for any of the sites (Table 10-7). However, proportions of the immediate biosynthetic products (18:1w7c and 16:1w7c) and their modification products (cyclopropyl and trans) differ between sites. Table 10-7 indicates that 18:1w7c is significantly higher in proportion for the Hardscrabble Creek profiles, followed by Swamp Gulch and the 20 cm depth horizons of Sand Coulee wetlands.

Figure 10-14 indicates that the cyclopropyl/cis ratios and trans/cis ratios also differ between sites, suggesting a difference in the microbial community physiological status. A full discussion of the lipid biochemistry and microbial physiology behind these proposed microbial "stress" indices is given in Guckert et al. (1986). The interpretation of the higher cyclopropyl/cis ratios at the Sand Coulee site must take into consideration the input of cyclopropyl PLFA from sulfate-reducing bacteria, where physiological response of cyclopropyl PLFA is unknown. It is evident that the Sand Coulee microbial communities are stressed in a way that might be analogous to anaerobic

microcosms which have been shown to have dramatic loss of cis-monoenoic PLFA with the corresponding production of cyclopropyl PLFA (Guckert et al. 1985).

The other dramatic physiological shift shown in Figure 10-14 is the extremely high trans/cis ratio in the Swamp Gulch microbial community. This ratio generally ranges from 0.01 to 0.08 for most bacterial cultures and 0.01 to 0.09 for marine, mangrove and estuarine sediment surfaces (Guckert et al. 1986). When Vibrio cholerae was starved for 7 days, the trans/cis ratio was 0.85 (Guckert et al. 1986). Significantly higher trans/cis ratios (0.15) were found in a highly consolidated sediment surrounding the burrow of a mud shrimp in estuarine sediment (Dobbs and Guckert 1988). Four hours after an estuarine sedimentary microbial community was disturbed by sieving, the trans/cis ratio increased to 0.95 (Findlay et al. 1988). It is, therefore, unusual to find an in situ trans/cis ratio above 0.50 as was observed in the Swamp Gulch sediments. This finding might be a response to low pH, metal toxicity, or another unknown stress. The differences between the trans/cis and cyclopropyl/cis stress of the Swamp Gulch and Sand Coulee wetlands are unknown. The control site, Hardscrabble Creek, is low for both "stress" lipid indices. It was previously suggested that this community might buffer AMD as the Swamp Gulch wetland has. It is unknown if the trans/cis ratio would follow the same pattern following AMD input.

The microbial populations at both Swamp Gulch and Sand Coulee wetlands are physiologically stressed. These physiological indices represent a means by which wetland performance can be measured. For example, if amendments applied to constructed wetlands resulted in improved AMD amelioration and the stress indices dropped in magnitude, this would indicate that the overall "health" of the microbial community improved. This would translate to enhanced acidity removal, sulfate control and metal immobilization.

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APPENDIX A

Analytical Methods and Quality Assurance and
Quality Control Statistics

Appendix A-1. Water Analysis Methods

Parameter	Method	Detection Limit	Units
<u>Physical Properties</u>			
Conductance, Specific @ 25°C	EPA 120.1	0.1	umhos/cm
Hardness, Total as CaCO ₃	EPA 130.2	1	mg/l
pH	EPA 150.1	0.1	Std. units
Residue: Total Filterable (TDS)	EPA 160.1	1	mg/l
<u>Metals</u>			
Aluminum	EPA 202.1/200.7	0.1	mg/l
Antimony	EPA 204.2	0.05	mg/l
Arsenic	EPA 206.2	0.005	mg/l
Cadmium	EPA 213.1/200.7	0.001	mg/l
Calcium	EPA 215.1/200.7	1	mg/l
Chromium	EPA 218.1/200.7	0.02	mg/l
Cobalt	EPA 219.1/200.7	0.01	mg/l
Copper	EPA 220.1/200.7	0.01	mg/l
Iron	EPA 236.1/200.7	0.03	mg/l
Lead	EPA 239.2/200.7	0.01	mg/l
Magnesium	EPA 242.1/200.7	1	mg/l
Manganese	EPA 243.1/200.7	0.02	mg/l
Nickel	EPA 249.1/200.7	0.03	mg/l
Potassium	EPA 258.1/200.7	1	mg/l
Selenium	EPA 270.2	0.005	mg/l
Silver	EPA 272.2/200.7	0.005	mg/l
Sodium	EPA 273.1/200.7	1	mg/l
Strontium	USGS 5 p. 150	0.1	mg/l
Zinc	EPA 289.1/200.7	0.01	mg/l

Appendix A-1. Continued.

Parameter	Method	Detection Limit	Units
<u>Inorganics, Non-Metallics</u>			
Alkalinity; Total, HCO ₃ , CO ₃ and OH	EPA 310.1	1	mg/l
Bromide	EPA 320.1	0.1	mg/l
Chloride	EPA 325.3	1	mg/l
Cyanide, total	EPA 335.2	0.005	mg/l
Fluoride, electrode	EPA 340.2	0.10	mg/l
Iodide	EPA 345.1	0.1	mg/l
Nitrogen: Nitrate plus Nitrite as N	EPA 353.2	0.05	mg/l
Phosphorus, ortho	EPA 365.1	0.01	mg/l
Sulfate	EPA 375.3	1	mg/l
<u>Organics</u>			
Organic Carbon, dissolved	EPA 415.1	2	mg/l

Procedures for calculating accuracy.

1. From the summarized information on the accuracy form the percent recovery of the blind field standard and the recovery of each lab spiked sample pair are calculated separately as:

$$\% \text{Recovery of BFS} = \frac{VA}{VK} \times 100; \text{ or } \% \text{Recovery of Spike} = \frac{SSR - SR}{SA} \times 100$$

where:

VA = analytical value of BFS
VK = known (or certified) value of BFS
SSR = spiked sample results
SR = sample results
SA = spike added

Perfect accuracy would be 100 percent recovery.

2. Calculate the standard deviation of all pairs.

$$SD = \left[\frac{(\text{Recovery}_i - \text{Recovery}_{\text{avg.}})^2}{n-1} \right]^{1/2}$$

where recovery_i is the individual recoveries, $\text{recovery}_{\text{avg.}}$ is the average recovery, and n is the number of values.

3. To validate recovery data, the individual recoveries are compared with the average recovery value to identify individual values that lie outside the range of reasonableness. Chauvenet's criterion is used to identify individual recovery values that lie outside this range.

To use Chauvenet's criterion, the screening variable is computed for recovery values that are suspected of lying outside the range of reasonableness.

$$\text{Screening Variable} = (\text{Recovery}_i - \text{Recovery}_{\text{avg.}}) / SD$$

The calculated screening variable is then compared to the maximum allowable value (Table B-1) for the appropriate number of recovery determinations. The suspect recovery value is set aside (set aside values are called "outliers") if the calculated screening variable equals or exceeds the maximum allowable value.

If outliers are identified using Chauvenet's criterion, a new average recovery and a new standard deviation are recalculated using the remaining "good" values, and Chauvenet's criterion is reapplied. This procedure is repeated until all surviving recovery values pass Chauvenet's criterion. (Usually one application and one recalculation are enough.) The final average recovery and final standard deviation are

calculated from the "surviving" recovery values. The final average recovery value is used to eliminate any bias from the laboratory data.

4. The range of uncertainty (R) in the recovery is then calculated.

$$\pm R = \pm tSD/(n)^{1/2}$$

where:

R is the range of uncertainty expressed as a percent

t is the value of the t distribution for the selected confidence level (90 percent) and (n-1) degrees of freedom

n is the number of samples

SD is the standard deviation.

The range of uncertainty, is used in conjunction with the average recovery to determine if bias adjustments are required.

5. Together, the final average recovery value for BFS and lab spike and the corresponding range of uncertainties constitute the QA statements of accuracy for a particular sampling program.
6. The completeness of accuracy data is that percentage of the total number of samples that remained after outliers are identified and set aside with Chauvenet's criterion.

Table B-1. Chauvenet's Criterion.

CHAUVENET'S CRITERION
FOR REJECTING A SUSPECTED VALUE^a

Number of Samples n	Maximum Allowable Values for (Recovery _i - Recovery _{avg}) / SD
3	1.901
4	1.983
5	2.015
6	2.111
7	2.164
8	2.195
9	2.214
10	2.228
11	2.279
12	2.318
13	2.348
14	2.373
15	2.393
16	2.409
17	2.424
18	2.435
19	2.445
20	2.454
21	2.462
22	2.469
23	2.475
24	2.480
25	2.485
26	2.502
27	2.517
28	2.530
29	2.543
30	2.555
40	2.634

^aBased on "t" distribution rather than the traditional "normal" distribution.

^bIndividual Recovery = Recovery_i
Average Recovery = Recovery_{avg}

Procedures for calculating precision.

1. From the summarized information on the precision form the relative percent difference (RPD) of each replicate pair (blind field replicates and laboratory duplicates) are calculated separately as:

$$RPD = \frac{|S-D|}{(S+D)/2} \times 100$$

where:

RPD = Relative Percent Difference

S = First sample value (original)

D = Second sample value (duplicate)

Perfect precision would result in 0% RPD.

2. Any RPD value exceeding the control limit of +20% is evaluated as an outlier using the Dixon's Q method. This method compares the difference between the suspected outlier and the value nearest to it in size with the difference between the highest and lowest values. The ratio of these differences (without regard to sign) is the Dixon's Q.

$$Q = \frac{|\text{suspect RPD value} - \text{nearest RPD value}|}{|\text{largest RPD value} - \text{smallest RPD value}|}$$

The critical values of Q for P=0.05 are given in Table B-2. If the calculated value of Q exceeds the critical value the suspect RPD value is rejected, and not used in the following precision QA statement calculations.

3. Calculate the relative standard deviation (RSD) of each duplicate pair (field duplicates and lab duplicates are treated separately).

$$RSD_{\text{each pair}} = SD/\text{Mean}$$

4. Calculate the RSD for all the pairs (field duplicates and lab duplicates are treated separately).

$$RSD_{\text{overall}} = \left[\sum_{i=1}^n \frac{RSD_{\text{each}}^2}{n-1_{\text{pairs}}} \right]^{1/2}$$

5. Calculate precision as a percent.

$$\text{Precision (\%)} = \frac{t_{n-1\text{pairs}} \times \text{RSD}_{\text{overall}}}{(n-1\text{pairs})^{1/2}} \times 100$$

where: t is value from the 90% probability level

6. Calculate the maximum uncertainty for any individual test result as:

$$x \pm x t \text{SD}_{\text{overall}} / (n-1\text{pairs})^{1/2}$$

where: x is the reported test result
t is the value of the t distribution at the 90% probability level.

7. The completeness of precision data is that percentage of the total number of RPD values that remained after outliers are identified and set aside with Dixon's Q ratio.

Table B-2. Critical values of Q^* ($P=0.05$)

<u>Sample size</u>	<u>Critical value</u>
4	0.831
5	0.717
6	0.621
7	0.570
8	0.524
9	0.492
10	0.464

*E.P. King. 1958. J. Am. Statisti. Assoc. Vol. 48, 531.

APPENDIX B

**Influent Flow Rates, Precipitation and Water Chemistry
Data from The Swamp Gulch Wetland**

Table B-1. Swamp Gulch daily flow (m3) at tailings dam flume.

Date	June	July	August	September	October	November
1		36.84	73.97	18.92	24.95	18.92
2		36.84	71.68	17.95	24.95	18.92
3		36.84	73.94	17.50	23.39	18.92
4		36.84	70.55	15.15	21.86	18.92
5		36.84	63.52	14.75	24.95	18.92
6		36.84	62.79	17.50	29.85	18.92
7		36.84	74.74	18.92	29.85	18.92
8		36.84	68.68	18.92	29.85	20.36
9		36.84	40.16	18.92	28.18	27.05
10		152.73	17.50	18.92	28.24	17.11
11		122.37	16.11	18.92	31.58	21.86
12		86.57	16.82	18.92	32.43	21.86
13		76.27	22.37	17.50	34.15	23.39
14		71.68	42.17	17.50	32.43	23.39
15	27.16	64.97	38.69	16.11	35.03	12.23
16	160.89	59.19	31.58	16.11	37.78	20.25
17	99.94	418.74	29.85	17.50	38.06	
18	171.62	731.79	28.18	18.92	26.99	
19	127.75	358.76	23.39	20.36	17.50	
20	78.90	238.74	21.86	20.36	16.11	
21	64.63	210.76	18.92	18.92	16.11	
22	57.43	213.56	18.92	18.92	16.11	
23	54.29	363.88	18.92	20.36	16.11	
24	53.27	191.33	19.88	20.36	16.79	
25	51.23	142.48	74.82	21.86	17.50	
26	45.00	120.47	38.23	20.36	18.92	
27	38.69	113.70	29.59	21.86	17.50	
28	36.84	111.04	29.57	23.39	18.92	
29	36.84	105.80	23.39	26.54	17.50	
30	36.84	98.13	22.37	26.54	16.79	
31		85.92	20.84		18.92	

Total	1141.32	4470.74	1204.00	578.78	759.32	319.93
Max	171.62	731.79*	74.82	26.54	38.06	27.05
Min	36.84	36.84	16.11	14.75	16.11	12.23*

Note: * Record maximum and minimum for the period

Appendix B-2. Precipitation at the Swamp Gulch wetland study.

Date	cm	in	Date	cm	in
6/09	0.25	0.10	12/02	0.36	0.14
6/10	0.25	0.10	12/04	0.20	0.08
6/16	1.02	0.40	12/08	0.18	0.07
6/17	0.30	0.12	12/09	0.25	0.10
6/18	1.27	0.50	12/11	1.52	0.60
6/21	0.13	0.05	- - - - -	- - - - -	- - - - -
TOTAL	3.22	1.27			
7/01	0.13	0.05	12/12/87-01/11/88		
7/05	0.13	0.05		4.24	1.67+
7/09	0.13	0.05	1/13	0.18	0.07
7/10	2.34	0.92	1/14	0.38	0.15
7/16	0.71	0.28	1/15	0.10	0.04
7/17	5.72	2.25	1/20	0.31	0.12
7/18	0.89	0.35	1/22	0.23	0.09
7/20	0.08	0.03	1/23	0.13	0.05
7/21	0.10	0.04	1/30	0.20	0.08
7/22	1.42	0.56	1/31	0.15	0.06
7/30	0.18	0.07	TOTAL	12/87 - 1/88	
TOTAL	11.83	4.65		5.92	2.33
8/07	0.20	0.08	2/01	0.05	0.02
8/10	0.13	0.05	2/02	0.18	0.07
8/14	0.71	0.28	2/03	0.48	0.19
8/15	0.25	0.10	2/04	0.20	0.08
8/16	0.10	0.04	2/06	0.05	0.02
8/24	0.05	0.02	2/07	0.64	0.25
8/25	1.07	0.42	2/08	1.35	0.53
8/26	0.03	0.01	2/09	0.28	0.11
8/27	0.03	0.01	2/10	0.08	0.03
8/28	0.05	0.02	2/11	0.10	0.04
TOTAL	2.62	1.03	2/12	0.30	0.12
9/27	0.08	0.03	2/13	0.05	0.02
9/30	0.10	0.04	2/14	0.41	0.16
TOTAL	0.18	0.07	2/15	0.66	0.26
10/15	0.08	0.03	2/17	0.05	0.02
10/16	0.38	0.15	2/21	0.76	0.30
TOTAL	0.46	0.18	2/22	0.13	0.05
11/02	0.13	0.05	TOTAL	5.77	2.27
11/03	0.20	0.08			
11/14	0.25	0.10			
11/19	1.07	0.42			
TOTAL	1.65	0.65			

Appendix B-3. Swamp Gulch wetland water chemistry data.

SITE	DATE	FIELD LAB	FIELD LAB	Ca	Mg	Na	K	HC03	S04	NO3	P	TDS	LAB	HARDNESS	SAR	CHARGE	Dissolved TSS	CN	
SAMPLED	PH	PH	TEMP, EC	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	calic- ulated 180 C					mg/l		
SWAMP 5L @ HINWAY 200	04/29/87	4.70	4.00	11.3	0.42	0.35	22	9	2.0	<1.0	0	166.0	0.22	<0.01	227.0	283.0	125.0	0.1	2.0
SWAMP 5L ABOVE MINE	04/29/87	7.77	6.80	8.9	0.09	0.09	10	4	1.0	<1.0	43	7.0	0.17	<0.01	44.5	64.0	41.5	0.1	-0.1
BLK FT RIV @ PASS CREEK	04/29/87	6.94	6.81	12.7	0.19	0.16	18	8	1.0	<1.0	55	29.0	0.13	<0.01	85.1	96.0	77.9	0.0	-0.6
BLK FT RIV BLW PAYMASTER	04/29/87	7.12	6.80	12.7	0.18	0.17	18	9	1.0	<1.0	51	35.0	0.16	<0.01	90.2	102.0	82.1	0.0	-0.8
BLK FT RIV @ MEADOW CRK	04/29/87	7.26		12.0	0.18							0.19	<0.01	1.1		0.1		0.2	
BLK FT R @ MEA CR RD BRG	04/29/87	7.31	6.80	11.3	0.17	0.16	17	8	1.0	<1.0	48	34.0	0.11	<0.01	85.6	103.0	75.5	0.1	-0.3
AAA-3-D	05/02/87	5.94	2.90	4.1	0.40	0.51	19	3	5.0	3.0	0	138.0	0.08	<0.01	202.6	241.0	129.5	0.2	-3.0
AAA-3-S	05/02/87	6.54	5.60	5.4	0.40	0.25	17	10	4.0	2.0	45	76.0	0.09	<0.01	162.4	212.0	134.9	0.2	-7.4
AA-2-D	05/02/87	6.62	4.20	5.5	0.60	0.28	15	8	8.0	4.0	0	122.0	0.13	0.020	174.7	274.0	97.1	0.4	-0.3
AA-2-S	05/02/87	6.25	4.60	4.7	0.32	0.22	14	7	10.0	3.0	2	85.0	0.13	<0.01	129.2	212.0	76.3	0.5	-2.1
A-3	05/02/87	5.01	3.60	4.9	0.34	0.30	15	8	5.0	4.0	0	119.0	0.13	<0.01	180.3	250.0	117.7	0.3	-4.1
B-2-D	05/02/87	4.47	3.40	7.1	0.37	0.31	13	7	4.0	4.0	0	117.0	0.12	<0.01	164.2	239.0	91.2	0.2	0.8
B-2-S	05/02/87	4.38	3.20	6.9	0.42	0.35	16	7	<1.0	<1.0	0	111.0	0.14	<0.01	161.0	246.0	111.4	0.0	-2.4
BC-1-D	05/01/87	6.44	5.70	6.1	0.30	0.23	20	8	8.0	3.0	18	72.0	0.16	<0.01	131.7	193.0	97.8	0.4	-4.8
BC-1-S	05/01/87	4.76	3.20	7.9	0.35	0.31	12	7	4.0	4.0	0	111.0	0.08	<0.01	157.9	234.0	90.5	0.2	0.0
BC-1-2	05/01/87	6.12	4.60	6.9	1.00	0.65	33	14	6.0	2.0	1	320.0	0.16	<0.01	483.1	551.0	319.2	0.2	-8.0
C-1-D	04/30/87	5.89	5.30	10.6	0.97	0.87	54	22	56.0	5.0	13	441.0	0.31	<0.01	642.8	572.0	318.8	1.6	-2.1
C-1-S	04/30/87	4.68	3.10	12.6	0.52	0.20	16	6	5.0	<1.0	0	218.0	0.19	<0.01	323.2	188.0	196.0	0.3	-5.5
C-1-5	04/30/87	6.16	5.30	8.9	0.58	0.36	26	10	7.0	<1.0	20	153.0	0.25	<0.01	280.4	220.0	227.2	0.3	-16.1
C-2-D	04/30/87	6.07	4.10	9.2	0.31	0.29	14	8	6.0	4.0	0	123.0	0.18	<0.01	0.0	198.0	0.0	0.0	0.0
C-2-S	04/30/87	3.98	3.40	12.4	0.31	0.31	13	7	5.0	4.0	0	117.0	0.15	<0.01	171.2	204.0	101.8	0.3	-1.6
C-3-D	04/30/87	7.12	7.20	7.9	0.70	0.53	1	<1	105.0	1.0	56	174.0	0.12	<0.01	323.1	367.0	23.2	21.4	-3.3
C-3-S	04/30/87	6.58	6.00	12.4	0.33	0.25	17	10	23.0	<1.0	143	6.0	0.13	<0.01	138.2	129.0	98.2	1.1	-3.8
C-4-D	04/30/87	6.37	6.10	12.6	0.29	0.25	22	10	7.0	<1.0	44	74.0	0.13	<0.01	141.4	158.0	101.8	0.3	-0.7
C-4-S	04/30/87	6.42	6.00	11.4	0.48	0.30	23	10	3.0	<1.0	147	24.0	0.37	0.03	159.3	184.0	130.8	0.4	-2.2
C-5-D	04/30/87	6.31	5.60	8.3	0.40	0.30	25	13	2.0	2.0	24	113.0	0.12	<0.01	199.9	185.0	164.9	0.1	-8.5
C-5-S	04/30/87	5.77	6.90	12.5	0.54	0.43	41	19	5.0	2.0	36	176.0	0.15	<0.01	266.9	307.0	184.7	0.2	1.4
CD-1-2	05/01/87	5.74	3.00	7.9	0.70	0.72	30	12	7.0	<1.0	0	224.0	0.19	<0.01	346.7	301.0	242.2	0.3	-9.9
CD-1	05/01/87	5.97	2.80	5.9	1.00	0.95	38	16	13.0	<1.0	0	316.0	0.09	<0.01	482.1	386.0	315.0	0.4	-9.3
D-1-D	04/30/87	6.44	6.10	6.6	0.54	0.34	39	17	5.0	1.0	184	24.0	0.23	<0.01	224.5	211.0	244.2	0.2	-14.7
D-1-S	05/01/87	6.07	5.60	5.8	0.89	0.62	66	24	16.0	<1.0	42	268.0	0.14	<0.01	445.1	458.0	317.0	0.4	-4.0
D-2-D	04/30/87	6.57	5.50	7.4	0.38	0.25	21	10	4.0	3.0	18	96.0	0.13	<0.01	170.4	181.0	138.2	0.2	-7.7
D-2-S	04/30/87	6.65	6.00	10.7	0.50	0.19	16	8	8.0	1.0	58	37.0	0.13	<0.01	109.1	125.0	83.3	0.4	-3.0
D-3	05/01/87	6.26	5.80	6.6	0.30	0.21	17	7	7.0	1.0	35	60.0	0.12	<0.01	122.4	151.0	87.6	0.4	-2.4
E-1	05/01/87	5.55	3.70	6.9	0.43	0.37	30	13	6.0	1.0	0	157.0	0.16	<0.01	229.3	223.0	156.9	0.2	-2.8

Appendix B-3. cont.

DATE	FIELD LAB	FIELD LAB	Ca	Mg	Na	K	HCO3	SO4	NO3	P	TDS	HARDNESS	SAR	CHARGE	Disolved TSS	CN				
SAMPLED	pH	TEMP.	EC	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	calc-	Carbon			mg/l					
		C	mmhos								ulated	180 C								
			/cm																	
05/01/87	6.56	6.20	5.7	0.61	0.34	35	14	6.0	<1.0	196	10.0	0.14	<0.01	172.9	195.0	153.7	0.2	0.0		
05/01/87	6.76	6.40	2.6	0.46	0.35	21	10	37.0	1.0	106	86.0	0.13	<0.01	211.9	219.0	93.7	1.7	-0.3		
05/01/87	6.27	3.70	3.8	0.32	0.27	19	10	3.0	3.0	0	111.0	0.10	<0.01	167.0	180.0	121.2	0.1	-4.2		
05/01/87	6.58	6.20	3.2	0.47	0.29	29	14	1.0	<1.0	141	24.0	0.11	0.02	163.6	146.0	164.6	0.0	-5.9		
05/01/87	6.25	6.10	5.2	0.32	0.27	26	14	2.0	1.0	27	99.0	0.14	<0.01	162.9	180.0	122.6	0.1	1.1		
05/01/87	4.60	2.70	4.6	2.59	2.31	266	95	6.0	2.0	0	1470.0	0.28	<0.01	1913.9	2000.0	1175.3	0.1	9.6		
05/01/87	3.56	2.90	4.8	2.32	1.95	151	60	4.0	2.0	0	1390.0	0.64	0.01	2571.0	1880.0	631.7	0.1	44.5		50
07/28/87	3.29	3.00	22.5	0.60	0.58	24	10	<1.0	<1.0	0	216.0	0.33	<0.01	278.1	366.0	134.8	0.0	7.0		
07/29/87	6.77	7.10	10.5	0.13	0.12	14	6	<1.0	1.0	69	9.0	0.19	<0.01	64.8	86.0	59.8	0.0	0.9		6
07/29/87	7.04	6.80	12.5	0.28	0.27	29	15	<1.0	<1.0	84	64.0	0.18	<0.01	151.7	163.0	134.3	0.0	0.0		<1
07/29/87	7.22	6.90	13.5	0.27	0.26	28	15	<1.0	<1.0	84	61.0	0.18	<0.01	147.6	180.0	131.8	0.0	-0.1		2
07/29/87	7.10	6.90	13.7	0.27	0.26	28	15	<1.0	<1.0	84	61.0	0.19	<0.01	148.1	196.0	131.8	0.0	0.0		2
07/29/87	7.28	6.90	13.9	0.27	0.25	28	14	1.0	1.0	80	60.0	0.18	<0.01	145.1	170.0	127.6	0.0	-0.2		<1
07/31/87	3.73	3.50	12.4	0.32	0.30	15	8	4.0	4.0	0	120.0	0.19	<0.01	175.4	243.0	108.9	0.2	-1.9		
07/31/87	6.31	5.40	15.1	0.32	0.18	10	6	2.0	4.0	21	54.0	0.11	0.02	106.0	124.0	80.1	0.1	-4.1		
07/31/87	4.68	4.20	12.0	0.36	0.31	13	7	19.0	4.0	0	130.0	0.10	<0.01	186.6	238.0	81.3	1.1	0.2		
07/31/87	5.79	4.60	15.4	0.22	0.16	12	7	6.0	4.0	2	59.0	0.19	0.02	95.4	159.0	65.4	0.3	-2.8		
07/31/87	5.19	3.40	22.5	0.29	0.29	12	7	3.0	4.0	0	116.0	0.10	<0.01	171.6	224.0	107.5	0.2	-2.5		
07/30/87	3.97	3.80	9.0	0.30	0.28	12	7	2.0	3.0	0	118.0	0.10	<0.01	160.9	232.0	88.0	0.1	2.2		
07/30/87	5.38	4.00	13.0	0.28	0.28	13	7	3.0	4.0	0	116.0	0.15	0.01	173.1	209.0	110.7	0.2	-2.9		
07/30/87	6.18	5.90	15.6	0.23	0.19	14	8	4.0	3.0	33	52.0	0.09	0.04	103.5	155.0	72.7	0.2	-0.2		<0.005
07/30/87	4.52	4.00	12.5	0.27	0.25	12	7	2.0	3.0	0	105.0	0.16	<0.01	149.2	213.0	90.7	0.1	-0.2	<2	
07/30/87	6.05	4.10	16.9	0.35	0.31	17	9	4.0	4.0	0	103.0	0.15	<0.01	166.5	198.0	126.1	0.2	-7.2		
07/28/87	6.37	5.60	16.3	0.79	0.80	49	23	11.0	4.0	33	288.0	0.20	<0.01	452.1	464.0	315.9	0.3	-6.6		
07/28/87	5.99	5.40	18.7	0.67	0.53	24	10	1.0	<1.0	16	255.0	0.16	<0.01	400.6	429.0	275.4	0.0	-9.6	3	<0.005
07/28/87	6.30	5.80	16.0	0.41	0.20	20	8	1.0	<1.0	99	53.0	0.15	<0.01	189.9	116.0	179.3	0.0	-12.9	5	<0.005
07/28/87	6.01	4.30	13.0	0.33	0.27	16	8	4.0	6.0	0	113.0	<0.05	<0.01	173.2	210.0	113.6	0.2	-3.6		
07/28/87	4.48	3.80	16.4	0.29	0.28	12	7	2.0	5.0	0	112.0	0.21	<0.01	182.3	217.0	96.8	0.9	-3.4		
07/29/87	5.90	3.80	12.8	0.33	0.28	12	7	15.0	5.0	0	120.0	0.13	<0.01	182.3	189.0	96.8	0.0	0.0		
07/29/87	6.68	6.00	15.5	0.45	0.24	25	13	5.0	1.0	155	2.0	0.14	<0.01	108.6	97.0	101.2	0.0	-1.4		
07/29/87	6.54	6.20	14.6	0.22	0.19	20	10	1.0	1.0	82	29.0	0.14	<0.01	108.6	97.0	101.2	0.0	-1.4		
07/29/87	6.04	6.10	17.8	0.34	0.25	23	12	3.0	2.0	151	2.0	0.14	<0.01	144.2	115.0	152.3	0.1	-7.6		
07/29/87	5.88	5.60	13.4	0.35	0.29	24	12	1.0	3.0	38	101.0	0.10	<0.01	189.2	224.0	152.1	0.0	-6.2		
07/30/87	6.29	6.20	15.0	0.39	0.33	32	15	2.0	2.0	85	92.0	0.18	<0.01	204.6	207.0	156.2	0.1	-2.4		
07/29/87	6.26	5.30	17.6	0.37	0.27	20	8	1.0	<1.0	28	96.0	0.13	<0.01	185.6	184.0	158.7	0.0	-11.0	2	<0.005

Appendix B-3. cont.

SITE	DATE SAMPLED	FIELD LAB pH	FIELD LAB TEMP. C	FIELD EC EC	FIELD LAB EC	Ca mg/l	Mg mg/l	Na mg/l	K mg/l	HC03 mg/l	SO4 mg/l	NO3 mg/l	P mg/l	TDS calculated	HARDNESS TDS	SAR	CHARGE Organic Carbon mg/l	TSS mg/l	CN	
CD-1	07/29/87	6.22	5.40	17.3	0.52	0.28	27	11	3.0	<1.0	38	93.0	0.08	<0.01	177.0	185.0	0.1	-5.0	5	<0.005
D-1-D	07/30/87	6.35	6.10	8.2	0.47	0.35	36	16	3.0	2.0	189	36.0	0.16	<0.01	227.8	212.0	0.1	-8.9		
D-1-S	07/30/87	6.33	5.80	11.2	0.70	0.70	76	26	10.0	<1.0	60	302.0	0.14	<0.01	510.1	530.0	0.3	-5.0	4	<0.005
D-2-D	07/30/87	6.47	6.00	10.4	0.40	0.28	24	12	3.0	4.0	92	66.0	0.18	<0.01	186.6	192.0	0.1	-6.2		
D-2-S	07/30/87	6.53	6.20	13.4	0.33	0.23	25	13	2.0	1.0	140	2.0	0.18	<0.01	133.6	131.0	0.1	-6.8		
D-3	07/30/87	6.46	6.20	12.7	0.31	0.16	16	9	1.0	2.0	99	2.0	0.12	<0.01	81.9	95.0	0.0	0.3		
E-1	07/30/87	6.08	5.90	12.6	0.39	0.28	26	12	3.0	1.0	57	83.0	0.13	<0.01	177.3	196.0	0.1	-5.3		
E-2	07/30/87	6.33	6.10	13.9	0.59	0.34	35	16	5.0	2.0	221	2.0	0.13	<0.01	202.1	197.0	0.2	-6.5		
E-3	07/30/87	6.45	6.40	12.3	0.46	0.31	31	16	6.0	1.0	126	63.0	0.14	<0.01	187.4	203.0	0.2	0.1		
F-1-D	07/30/87	6.24	5.90	10.4	0.58	0.44	42	19	8.0	1.0	116	122.0	0.15	0.04	289.8	304.0	0.3	-8.5	288	
F-1-S	07/30/87	6.22		11.3	0.41									9.8	12.4		-4.5			
F-2-D	07/30/87	5.81	4.50	11.0	0.29	0.24	18	10	1.0	4.0	0	102.0	0.16	<0.01	190.0					
F-2-S	07/30/87	6.10	5.90	11.8	0.33	0.24	23	13	4.0	1.0	92	35.0	0.13	0.02	132.8	157.0	0.2	-4.3		
F-3	07/30/87	5.93	6.00	14.9	0.27	0.25	26	14	<1.0	2.0	40	89.0	0.16	<0.01	152.8	175.0	0.0	0.4		
SEEP-D	07/29/87	4.20	3.40	17.4	2.06	1.93	229	92	9.0	2.0	0	1230.0	0.33	<0.01	1820.0				869	
SEEP-S	07/29/87	3.33	2.90	20.9	1.89	1.70	133	54	<1.0	3.0	0	1110.0	0.20	<0.01	1332.6	2120.0	0.0	23.7	1010	
AAA-3-SURF	07/31/87	6.80	7.00	15.9	0.23	0.23	26	12	1.0	<1.0	91	42.0	0.11	<0.01	127.7	153.0	0.0	0.4	<1	
BC-1-SURF	07/31/87	6.11	6.20	17.1	0.29	0.26	28	8	8.0	4.0	59	61.0	0.13	<0.01	156.7	208.0	0.3	-3.3	53	
C-1.5-SURF	07/31/87	2.93	2.90	16.4	0.80	0.70	20	8	3.0	2.0	0	217.0	0.13	<0.01	287.5	326.0	0.1	5.0	92	
C-3-SURF	07/31/87	6.44	6.50	17.4	0.27	0.23	27	13	1.0	1.0	115	28.0	0.09	0.02	135.1	154.0	0.0	-2.4	12	
CD-1-SURF	07/31/87	2.90	2.90	15.4	0.89	0.84	49	14	4.0	1.0	0	299.0	0.13	<0.01	397.4	440.0	0.1	5.5	24	
F-1-SURF	07/31/87	3.50	3.30	19.4	0.35	0.44	26	11	5.0	1.0	0	142.0	0.11	<0.01	208.7	255.0	0.2	-2.1	336	
SWAMP 6L @ HIWAY 200	01/11/88	3.88	2.80	0.7	0.53	0.65	22	9	4.0	1.0	0	225.0	0.69	<0.01	317.2	247.0	0.2	0.2	213	
SWAMP 6L ABOVE MINE	01/11/88	7.25	6.20	0.5	0.14	0.12	14	6	3.0	1.0	55	16.0	0.62	<0.01	68.5	73.0	0.2	-0.7	98	
BLK FT RIV @ PASS CRE	01/12/88	6.60	6.30	0.8	0.29	0.31	35	18	2.0	1.0	98	74.0	0.60	<0.01	180.1	191.0	0.1	-0.9	83	
BLK FT R @ MEA CR RD BR	01/12/88	6.70	6.10	0.3	0.24	0.27	29	15	3.0	1.0	48	87.0	0.58	<0.01	161.3	176.0	0.1	-1.2	3	
BLK FT R @ MEA CR RD BR6	01/12/88	6.70	6.20	0.3	0.24	0.27	29	15	3.0	1.0	50	88.0	0.57	<0.01	162.8	160.0	0.1	-1.0	3	
AA-2-D	01/13/88	5.02	3.40	-5.5	0.36	0.32	22	9	21.0	8.0	0	130.0	0.68	<0.01	200.7	231.0	1.0	-3.6		
AA-2-S	01/13/88	6.04	4.74	4.1	0.22	0.15	13	7	8.0	5.0	2	60.0	0.65	0.01	104.3	145.0	0.4	-3.6		
B-2-D	01/12/88	4.25	3.30	5.3	0.18	0.32	13	7	5.0	4.0	0	128.0	0.77	<0.01	176.2	172.0	0.3	2.3		
B-2-S	01/12/88	4.50	3.10	5.0	0.25	0.29	13	7	5.0	5.0	0	124.0	0.20	<0.01	175.5	179.0	0.3	0.7		
BC-1-D	01/12/88	6.20	5.70	1.0	0.20	0.20	16	9	8.0	7.0	29	62.0	0.53	0.02	120.7	155.0	0.4	-1.1		
BC-1-S	01/11/88	5.30	4.20	1.5	0.22	0.21	10	6	5.0	4.0	156	89.0	1.27	<0.01	210.7	156.0	0.3	13.0		
BC-1-2	01/11/88	6.30	4.90	1.0	0.33	0.22	15	7	6.0	4.0	7	79.0	0.96	<0.01	158.2	142.0	0.3	-13.7		
C-1-D	01/11/88	6.75	5.60	3.7	0.56	0.42	36	17	10.0	5.0	70	140.0	0.72	<0.01	288.7	257.0	0.3	-11.0		

Appendix B-3. cont.

SITE	DATE SAMPLED	FIELD LAB pH	FIELD LAB TEMP. C	FIELD EC EC	LAB EC EC	Ca mg/l	Mg mg/l	Na mg/l	K mg/l	HCO3 mg/l	SO4 mg/l	NO3 mg/l	P mg/l	TDS calc- ulated 180 C	HARDNESS	SAR	CHARGE Organic Carbon mg/l	Disolved TSS mg/l	CN mg/l
C-1-S	01/11/88	6.28	4.10	1.1	0.39	0.35	20	8	5.0	1.0	0	174.0	0.86	<0.01	259.6	240.0	166.9	0.2	-5.0
C-1-S	01/12/88	6.39	5.30	2.0	0.31	0.26	23	10	4.0	<1.0	39	87.0	0.67	<0.01	196.7	128.0	183.9	0.2	-16.0
C-2-D	01/12/88	3.49	3.20	0.5	0.30	0.32	20	10	6.0	7.0	0	137.0	0.68	<0.01	198.3	207.0	116.2	0.3	-0.1
C-2-S	01/12/88	4.27	3.20	3.5	0.24	0.31	15	8	5.0	5.0	0	120.0	0.67	<0.01	176.9	191.0	107.3	0.3	-1.5
C-4-D	01/13/88	6.13	6.20	0.0	0.28	0.25	28	14	6.0	1.0	61	75.0	0.59	<0.01	161.3	154.0	130.0	0.2	-2.5
C-4-S	01/13/87	5.78	5.70	0.0	0.28	0.16	14	8	3.0	2.0	109	21.0	0.66	<0.01	131.3	102.0	116.4	0.2	-5.0
CD-1	01/12/88	6.19	4.90	2.0	0.41	0.37	36	13	5.0	<1.0	10	183.0	0.66	<0.01	296.4	266.0	223.5	0.2	-9.9
CD-1-2	01/12/88	6.00	4.70	3.0	0.30	0.19	21	9	3.0	<1.0	37	85.0	0.69	<0.01	181.1	128.0	158.3	0.1	-11.9
D-1-D	01/12/88	6.29	6.20	6.0	0.39	0.30	36	16	5.0	2.0	163	37.0	0.67	0.02	212.9	163.0	213.6	0.2	-10.4
D-1-S	01/12/88	6.31	5.90	2.0	0.51	0.40	47	18	8.0	<1.0	134	98.0	0.68	<0.01	287.6	222.0	262.4	0.3	-12.3
E-1	01/13/88	5.80	4.20	3.1	0.49	0.41	38	16	7.0	1.0	0	186.0	0.68	<0.01	292.2	260.0	221.8	0.2	-9.1
D-2-D	01/12/88	6.15	6.00	3.0	0.32	0.26	27	13	7.0	3.0	127	48.0	0.61	<0.01	177.3	138.0	144.8	0.3	-2.4
D-2-S	01/12/88	6.40	6.00	2.5	0.32	0.25	22	13	5.0	1.0	123	24.0	0.65	<0.01	155.1	101.0	151.8	0.2	-8.4
E-2	01/13/88	6.58	6.00	1.5	0.39	0.28	28	14	7.0	1.0	169	5.0	0.69	<0.01	175.1	148.0	176.3	0.3	-10.4
F-1-D	01/13/88	5.75	6.30	5.4	0.50	0.37	41	20	8.0	1.0	189	40.0	0.67	<0.01	235.9	206.0	235.2	0.3	-10.0
F-1-S	01/13/88	6.50	6.50	3.0	0.47	0.34	38	19	7.0	<1.0	220	4.0	0.68	<0.01	205.4	187.0	210.5	0.2	-7.4
F-2-D	01/13/88	5.73	3.40	3.0	0.31	0.29	17	9	4.0	3.0	0	115.0	0.76	<0.01	169.8	172.0	112.6	0.2	-2.6
F-2-S	01/13/88	6.50	6.50	3.0	0.31	0.34	38	19	7.0	<1.0	220	4.0	0.68	<0.01	205.4	186.0	210.5	0.2	-7.4
SEEP-D	01/13/88	3.74	3.00	3.7	2.04	1.73	192	77	9.0	2.0	0	938.0	0.16	<0.01	1281.2	1370.0	888.5	0.1	0.2
SEEP-S	01/13/88	3.82		3.5											9.6		0.0		-2.7

Appendix B-3. cont.

SITE	DATE SAMPLED	Cl	F	Br	I	Fe	Mn	Cu	Pb	Zn	Cd	Ni	Se	Ag	Al	As	Co	Cr	Sb	Sr
		mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
SWAMP EL @ HWAY 200	04/29/87	1.0	0.25			18.40	3.88	0.79		2.84	0.038	0.04	<0.005	<0.005	6.100	<0.005	<0.010	<0.020	0.010	
SWAMP EL ABOVE MINE	04/29/87	<1.0	<0.10			0.05	<0.02	<0.01		<0.01	<0.001	<0.03	0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
BLK FT RIV @ PASS CREEK	04/29/87	1.0	<0.10			0.04	0.03	<0.01		0.26	<0.001	<0.03	0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
BLK FT RIV BLW PAYMASTER	04/29/87	1.0	<0.10			0.06	0.04	<0.01		0.28	<0.001	<0.03	0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
BLK FT RIV @ MEADOW CRK	04/29/87		0.50			0.06	0.04	<0.01		0.27	<0.001	<0.03	0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
BLK FT R @ MEA CR RD BR6	04/29/87	1.0	<0.10			0.06	0.04	<0.01		0.24	<0.001	<0.03	0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
AAA-3-D	05/02/87	1.0	0.05			27.40	0.89	<0.01		0.14	<0.001	<0.03	<0.005	<0.005	<0.100	0.021	<0.010	<0.020	0.010	
AAA-3-S	05/02/87	1.0	0.49			28.60	1.04	<0.01		0.03	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.010	
AA-2-D	05/02/87	1.0	0.49			14.90	0.80	0.02		0.35	0.003	<0.03	<0.005	<0.005	1.700	0.005	<0.010	<0.020	<0.010	
AA-2-S	05/02/87	1.0	0.47			7.02	0.59	<0.01		<0.01	<0.001	<0.03	<0.005	<0.005	0.200	<0.005	<0.010	<0.020	<0.010	
A-J	05/02/87	1.0	0.45			26.40	1.22	<0.01		0.06	<0.001	<0.03	<0.005	<0.005	0.200	<0.005	<0.010	<0.020	0.010	
B-2-D	05/02/87	1.0	0.40			16.70	0.64	0.07		0.29	<0.001	<0.03	<0.005	<0.005	3.000	<0.005	<0.010	<0.020	<0.010	
B-2-S	05/02/87	1.0	0.40			23.80	0.63	<0.01		<0.01	<0.001	<0.03	<0.005	<0.005	0.600	<0.005	<0.010	<0.020	0.010	
BC-1-D	05/01/87	2.0	0.24			8.33	0.86	0.01		0.21	0.002	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
BC-1-S	05/01/87	1.0	0.41			17.70	0.61	<0.01		0.05	<0.001	<0.03	<0.005	<0.005	2.800	<0.005	<0.010	<0.020	<0.010	
BC-1-2	05/01/87	5.0	0.19			145.00	2.19	<0.01		0.07	0.003	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.030	
C-1-D	04/30/87	1.0	0.15			52.10	4.51	<0.01		0.30	0.008	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.030	
C-1-S	04/30/87	1.0	0.15			73.30	2.57	<0.01		0.44	0.003	<0.03	<0.005	<0.005	0.900	<0.005	<0.010	<0.020	0.010	
C-1-3	04/30/87	2.0	0.30			67.60	3.89	<0.01		0.02	0.007	<0.03	<0.005	<0.005	0.100	<0.005	<0.010	<0.020	0.020	
C-2-D	04/30/87	1.0	0.21			23.30	1.08	<0.01		0.14	0.088	<0.03	<0.005	<0.005	0.200	<0.005	<0.010	<0.020	0.010	
C-2-S	04/30/87	1.0	0.49			22.60	0.68	<0.01		0.24	0.003	<0.03	<0.005	<0.005	0.900	<0.005	<0.010	<0.020	0.010	
C-3-D	04/30/87	3.0	0.28			10.40	0.21	<0.01		0.03	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.010	
C-3-S	04/30/87	2.0	0.82			8.13	0.24	<0.01		<0.01	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
C-4-D	04/30/87	2.0	0.17			3.16	0.78	<0.01		<0.01	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
C-4-S	04/30/87	2.0	0.28			18.00	0.81	<0.01		<0.01	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
C-5-D	04/30/87	1.0	0.18			27.30	4.42	<0.01		0.02	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	0.020	<0.020	0.020	
C-5-S	04/30/87	1.0	0.16			2.30	13.30	<0.01		1.18	0.007	<0.03	<0.005	<0.005	<0.100	<0.005	0.090	<0.020	0.030	
CD-1-2	05/01/87	2.0	0.10			65.80	5.06	<0.01		0.05	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.020	
CD-1	05/01/87	5.0	<0.10			86.10	7.26	<0.01		0.04	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.020	
D-1-D	04/30/87	2.0	0.25			42.90	2.58	<0.01		0.02	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.020	
D-1-S	05/01/87	18.0	0.10			29.80	13.50	<0.01		0.49	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.030	
D-2-D	04/30/87	1.0	0.46			24.90	1.07	<0.01		<0.01	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.010	
D-2-S	04/30/87	2.0	0.44			5.84	2.08	<0.01		0.02	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
E-1	05/01/87	3.0	0.25			9.12	0.68	<0.01		0.04	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
E-1-1	05/01/87	3.0	0.15			15.90	2.55	<0.01		0.11	0.002	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.020	
E-1-2	05/01/87	3.0	0.37			0.23	5.26	<0.01		0.03	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
E-1-3	05/01/87	1.0	0.17			0.04	3.32	<0.01		0.12	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	

Appendix B-3. cont.

SITE	DATE SAMPLED	Cl mg/l	F mg/l	Br mg/l	I mg/l	Fe mg/l	Mn mg/l	Cu mg/l	Pb mg/l	Zn mg/l	Cd mg/l	Ni mg/l	Se mg/l	Ag mg/l	Al mg/l	As mg/l	Co mg/l	Cr mg/l	Sb mg/l	Sr mg/l
F-2-D	05/01/87	1.0	0.38			18.20	1.15	<0.01	<0.01	0.17	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.010	
F-2-S	05/01/87	3.0	0.29			19.30	3.20	<0.01	<0.01	0.02	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	0.020	
F-3	05/01/87	7.0	0.21			<0.03	0.03	<0.01	<0.01	0.20	<0.001	<0.03	<0.005	<0.005	<0.100	<0.005	<0.010	<0.020	<0.010	
SEEP-D	05/01/87	2.0	0.78			57.80	68.10	1.41	1.41	12.10	0.464	0.15	<0.005	0.015	7.700	<0.005	0.170	<0.020	0.090	
SEEP-S	05/01/87	3.0	0.73			4.28	42.40	14.30	14.30	39.90	0.538	0.40	<0.005	0.009	7.950	0.044	0.180	<0.020	0.110	
SWAMP G @ HINWAY 200	07/28/87	<1.0	0.33			18.80	3.92	0.67	0.67	2.43	0.029	0.04	<0.005	<0.005	7.100	<0.005	<0.010	<0.010	<0.010	
SWAMP GL ABOVE MINE	07/29/87	<1.0	<0.10			0.06	<0.02	<0.01	<0.01	<0.01	<0.001				<0.100					
BLK FT RIV @ PASS CREEK	07/29/87	<1.0	<0.10			0.06	0.08	<0.01	<0.01	0.56	0.001				<0.100					
BLK FT R BLW PAYMASTER	07/29/87	<1.0	<0.10			0.06	0.07	<0.01	<0.01	0.45	<0.001				<0.100					
BLK FT RIV @ MEADOW CRK	07/29/87	1.0	<0.10			0.06	0.08	<0.01	<0.01	0.44	<0.001				<0.100					
BLK FT R @ MEA CR RD BRG	07/29/87	1.0	<0.10			0.04	0.08	<0.01	<0.01	0.39	<0.001				<0.100					
AAA-3-D	07/31/87	1.0	0.39	<0.100	<0.100	21.50	0.90	<0.01	<0.01	0.39	0.003									
AAA-3-S	07/31/87	1.0	0.62	<0.100	<0.100	17.00	0.78	<0.01	<0.01	0.13	0.001									
AA-2-D	07/31/87	1.0	0.35			11.20	0.64	<0.01	<0.01	0.33	0.012									
AA-2-S	07/31/87	1.0	0.93			3.70	0.39	0.02	0.02	0.18	<0.001								0.500	
A-3	07/31/87	1.0	0.46			27.20	0.73	<0.01	<0.01	0.14	0.001								0.200	
B-2-D	07/30/87	1.0	0.59			16.30	0.62	<0.01	<0.01	0.26	0.001								2.900	
B-2-S	07/30/87	1.0	0.62	<0.100	<0.100	27.60	0.71	<0.01	<0.01	0.03	<0.001								0.300	
BC-1-D	07/30/87	2.0	0.64			2.68	0.80	<0.01	<0.01	0.05	0.010								<0.100	
BC-1-S	07/30/87	1.0	0.62	<0.100	<0.100	17.80	0.62	<0.01	<0.01	0.02	<0.001								2.100	
BC-1-2	07/30/87	2.0	0.27			26.00	1.00	<0.01	<0.01	0.05	0.008								<0.100	
C-1-D	07/28/87	1.0	0.39			55.20	4.10	<0.01	<0.01	0.02	0.001								<0.100	
C-1-S	07/28/87	<1.0	0.14		<0.100	97.30	4.08	<0.01	<0.01	0.02	0.004								<0.100	
C-1.5	07/28/87	1.0	0.50	<0.100	<0.100	53.80	3.19	0.02	0.02	<0.01	0.002								<0.100	
C-2-D	07/28/87	2.0	0.36	<0.100	<0.100	22.70	1.02	<0.01	<0.01	0.06	0.002								<0.100	
C-2-S	07/28/87	1.0	0.64	<0.100		21.30	0.62	<0.01	<0.01	0.25	0.002								0.700	
C-3-D	07/29/87	1.0	0.38			21.20	0.49	<0.01	<0.01	0.13	<0.001								0.100	
C-3-S	07/29/87	1.0	0.67			20.90	0.35	<0.01	<0.01	<0.01	<0.001								<0.100	
C-4-D	07/29/87	1.0	0.26			5.64	0.23	<0.01	<0.01	<0.01	<0.001								<0.100	
C-4-S	07/29/87	1.0	0.50			25.40	0.87	<0.01	<0.01	<0.01	<0.001								<0.100	
C-5-D	07/29/87	1.0	0.30			23.90	4.15	<0.01	<0.01	0.02	<0.001								<0.100	
C-5-S	07/30/87	1.0	0.46			8.10	13.10	<0.01	<0.01	0.05	<0.001								<0.100	
CD-1-2	07/29/87	<1.0	0.31	<0.100	<0.100	42.30	3.11	<0.01	<0.01	0.01	<0.001								<0.100	
CD-1	07/29/87	3.0	0.22	0.100	<0.100	14.80	5.68	<0.01	<0.01	0.05	<0.001								<0.100	
D-1-D	07/30/87	1.0	0.36			37.80	2.53	<0.01	<0.01	0.01	<0.001								<0.100	
D-1-S	07/30/87	16.0	0.28	0.200	<0.100	39.60	14.80	<0.01	<0.01	0.04	0.002								<0.100	
D-2-D	07/30/87	2.0	0.70			28.20	1.27	<0.01	<0.01	0.04	<0.001								<0.100	

Appendix B-3. cont.

SITE	DATE SAMPLED	Cl	F	Br	I	Fe	Mn	Cu	Pb	Zn	Cd	Ni	Se	Ag	Al	As	Co	Cr	Sb	Sr
B-2-S	07/30/87	2.0	0.62			17.30	1.68		<0.01		0.02	<0.001			<0.100					
B-3	07/30/87	1.0	0.55			0.95	0.62		<0.01		0.02	<0.001			<0.100					
E-1	07/30/87	2.0	0.36			18.40	3.33		<0.01		0.02	<0.001			<0.100					
E-2	07/30/87	2.0	0.64			23.60	7.03		<0.01		0.01	<0.001			<0.100					
E-3	07/30/87	1.0	0.22			0.46	6.55		<0.01		0.05	<0.001			<0.100					
F-1-D	07/30/87	2.0	0.19			34.70	3.65		<0.01		0.04	<0.001	<0.03		<0.100					
F-1-S	07/30/87					6.93	2.74		<0.01		0.07	<0.001	0.03	<.005	<0.100	<.005	<.010		<.010	0.010
F-2-D	07/30/87	1.0	0.55			13.20	1.16		<0.01		0.29	<0.001			<0.100					
F-2-S	07/30/87	1.0	0.41			7.91	2.00		<0.01		0.06	<0.001			<0.100					
F-3	07/30/87	1.0	0.30			<0.03	<0.02		<0.01		0.15	<0.001			<0.100					
SEEP-D	07/29/87	3.0	0.21			63.20	62.10		0.35	0.12	4.89	0.190	0.09		2.000					
SEEP-S	07/29/87	2.0	1.00			5.83	37.50		12.30	1.02	25.90	0.449	0.35		61.300					
AAA-3-SURF	07/31/87	1.0	0.05	0.100	<.100	0.07	0.03		<0.01	<0.01	0.21	<0.001	<0.03		<0.100					<.010
BC-1-SURF	07/31/87	8.0	0.36	0.200	<.100	8.95	1.18		<0.01	<0.01	0.03	<0.001	<0.03		<0.100					<.010
C-1.5-SURF	07/31/87	1.0	0.20	<.100	<.100	32.50	2.86		0.02	0.09	0.63	0.003	<0.03		2.600					<.010
C-3-SURF	07/31/87	1.0	0.19	<.100	<.100	6.94	0.24		<0.01	<0.01	0.08	<0.001	<0.03		<0.100					<.010
CD-1-SURF	07/31/87	2.0	0.20	<.100	<.100	22.10	4.83		<0.01	0.09	1.02	0.003	<0.03		1.800					<.010
F-1-SURF	07/31/87	4.0	0.05	<.100	<.100	15.10	4.30		0.04	<0.01	0.12	<0.001	<0.03		0.400					<.010
SWAMP 5L @ HWAY 200	01/11/88	3.0	0.23			48.40	3.38		0.37	0.04	0.024	0.03		<.005	5.500	<.005	<.010	<.020	<.050	
SWAMP 5L ABOVE MINE	01/11/88	<1.0	<.10			0.11	0.21		<0.01	<0.01	0.002				<0.100			<.020		
BLK FT RIV @ PASS CRE	01/12/88	1.0	<.10			0.01	0.21		<0.01	<0.01	0.003				<0.100					
BLK FT R @ MEA CR RD BR	01/12/88	1.0	0.10			0.15	0.37		<0.01	<0.01	<0.001				<0.100					
BLK FT R @ MEA CR RD BR6	01/12/88	1.0	0.10			0.15	0.36		<0.01	<0.01	<0.001				<0.100					
AA-2-D	01/13/88	1.0	0.17			8.31	0.51		<0.01	<0.01	0.007				0.200					
AA-2-S	01/13/88	4.0	0.52			4.64	0.46		<0.01	<0.01	<0.001				0.300					
B-2-D	01/12/88	1.0	0.46			16.30	0.63		<0.01	<0.01	<0.001				3.000					
B-2-S	01/12/88	1.0	0.50			19.10	0.64		<0.01	<0.01	<0.001				1.200					
BC-1-D	01/12/88	2.0	0.63			0.66	0.55		<0.01	<0.01	<0.001				0.700					
BC-1-S	01/11/88	1.0	0.26			16.90	0.50		0.03	0.03	0.044				0.600					
BC-1-2	01/11/88	3.0	0.30			38.50	0.96		0.03	0.03	0.032				<0.100					
C-1-D	01/11/88	1.0	0.46			41.30	2.81		<0.01	<0.01	0.002				<0.100					
C-1-S	01/11/88	1.0	0.30			46.90	2.56		<0.01	<0.01	0.002				0.400					
C-1.5	01/12/88	1.0	0.28			47.60	3.43		<0.01	<0.01	<0.001				<0.100					
C-2-D	01/12/88	2.0	0.23			14.00	1.34		<0.01	<0.01	0.004				0.500					
C-2-S	01/12/88	1.0	0.92			20.60	0.70		<0.01	<0.01	<0.001				0.800					
C-4-D	01/13/88	1.0	0.14			1.36	0.22		<0.01	<0.01	<0.001				<0.100					
C-4-S	01/13/87	1.0	0.26			27.10	0.69		<0.01	<0.01	<0.001				<0.100					

Appendix B-3. cont.

SITE	DATE SAMPLED	Cl mg/l	F mg/l	Br mg/l	I mg/l	Fe mg/l	Mn mg/l	Cu mg/l	Pb mg/l	Zn mg/l	Cd mg/l	Ni mg/l	Se mg/l	Ag mg/l	Al mg/l	As mg/l	Co mg/l	Cr mg/l	Sb mg/l	Sr mg/l
CD-1	01/12/88	2.0	0.11			44.70	6.49		<0.01		0.001				<0.100					
CD-1-2	01/12/88	1.0	0.15			38.40	4.12		<0.01		0.002				0.100					
D-1-D	01/12/88	1.0	0.30			32.30	2.41		<0.01		<0.001				<0.100					
D-1-S	01/12/88	4.0	0.23			39.60	5.73		<0.01		<0.001				<0.100					
E-1	01/13/88	4.0	0.25			34.10	5.14		<0.01		<0.001				0.100					
D-2-D	01/12/88	1.0	0.52			13.30	1.40		<0.01		<0.001				<0.100					
D-2-S	01/12/88	1.0	0.55			24.20	3.18		<0.01		<0.001				<0.100					
E-2	01/13/88	2.0	0.65			27.20	6.44		<0.01		<0.001				<0.100					
F-1-D	01/13/88	1.0	0.26			28.20	2.82		<0.01		<0.001				<0.100					
F-1-S	01/13/88	3.0	0.31			20.90	3.80		<0.01		<0.001				<0.100					
F-2-D	01/13/88	1.0	0.44			18.50	1.09		<0.01		<0.001				0.100					
F-2-S	01/13/88	3.0	0.31			20.90	3.80		<0.01		<0.001				0.100					
SEEP-D	01/13/88	1.0	<0.01			51.40	53.10		0.18	0.27	0.053	0.06	0.009	0.005	1.200	<0.005	0.220		<0.050	
SEEP-S	01/13/88								9.27			0.34		0.005		0.018	0.140		0.050	

Appendix B-4. Elevation of sample stations at the Swamp Gulch wetland site.

Site	Ground Elevation	Elevation TOC1	WS2 Elevation 4/29/87	WS Elevation 6/3/87	WS Elevation 7/27/87	Potentimetric WS 8/13/87	WS Elevation 8/13/87	WS Elevation 1/13/88
Meters								
culvert bottom	1577.22			1578.04				
AAA-3-D	1578.39	1579.22	1578.57		1578.53			
AA-3-S	1578.38	1579.16	1578.48		1578.47			1578.81
AA-2-D	1578.17	1579.10	1578.85		1578.86			1578.24
AA-2-S	1578.18	1578.13	1578.40		1578.41			
A-3	1577.24	1578.13	1577.46		1577.53			
B-2-D	1577.31	1578.14	1577.52		1577.54			1577.58
B-2-S	1577.32	1578.14	1577.32		1577.33		1577.56	1577.35
BC-1-D	1577.78	1578.58	1577.46		1577.94		1577.32	1577.72
BC-1-S	1577.77	1578.50	1577.76		1577.83			1577.72
BC-1-2	1577.43	1578.21	1577.36		1577.39			1577.64
C-1-D	1577.82	1578.46			1577.77		1577.45	1577.38
C-1-S	1577.75	1578.42	1577.91		1577.88		1577.68	1577.72
C-1-5	1577.02	1577.80	1577.00		1576.99		1576.97	1577.01
C-2-D	1576.21	1577.06	1576.89		1576.87		1576.84	1576.97
C-2-S	1576.24	1577.06	1576.37		1576.32		1576.32	1576.33
C-3-D	1575.94	1576.97	1576.09		1576.08			
C-3-S	1575.96	1576.99	1576.07		1576.08			
C-4-D	1575.58	1576.36	1575.54		1575.51			
C-4-S	1575.49	1576.37	1575.55		1575.55			1575.30
C-5-D	1575.24	1576.04	1574.71		1574.57			1575.25
C-5-S	1575.18	1575.98	1574.68		1574.56			
CD-1-2	1576.88	1577.68	1576.87		1576.88		1576.76	1576.82
CD-1	1576.83	1577.62	1576.79		1576.81		1576.74	1576.80
D-1-D	1575.93	1576.75	1576.11		1576.04		1576.00	1575.99
D-1-S	1575.90	1576.73	1575.84		1575.91		1575.90	1575.96
D-2-D	1575.68	1576.52	1575.76		1575.72		1575.71	1575.71
D-2-S	1575.68	1576.49	1575.73		1575.73		1575.73	1575.74
D-3		1575.99	1575.19		1575.21			
E-1	1574.30	1575.10	1574.33		1574.32		1574.31	1574.38
E-2	1574.31	1575.10	1574.24		1574.22		1574.20	1574.25
E-3	1573.78	1574.72	1573.60		1573.51			
F-1-D	1573.13	1573.97					1573.12	1573.13
F-1-S	1573.13	1573.98					1573.11	
F-2-D	1572.64	1573.61	1572.50		1572.43			1572.43
F-2-S	1572.59	1573.56	1572.54		1572.47			1572.44
F-3	1572.98	1573.80	1572.55		1572.47			
SEEP-D	1580.14	1580.95	1579.98		1579.92			1579.65
SEEP-S	1580.15	1580.95	1580.06		1579.98			1579.65

1 Top Of Casing
2 Water Surface

APPENDIX C

Chemical and Physical Data from the Swamp Gulch Wetland Sediments

Appendix C-1. Swamp Gulch wetland sediment chemistry data.

NO	SITE	DEPTH HIGH	DEPTH LOW	DEPTH PK	CEC meq/ 100g	AEC meq/ 100g	EC mmhos /cm	ORGAN CARB %	S %	Si %	C %	TEXT URE	HCO3 mg/l	Cl mg/l	F ppm	SO4 mg/l	Ca ppm	K ppm	Mg ppm	Na ppm	P ppm
1	AAA-3-AH	0.00	0.24	4.9			3.41								0.00	2750					
2	AAA-3-AH	0.24	0.61	4.3			0.76								0.00	330					
3	AAA-3-AH	0.19	1.22	4.1			0.26								0.00	70					
4	AAA-3-D	1.46	2.13	3.8			1.18								0.00	490					
5	AAA-3-D	2.13	2.44	3.9			0.49								0.00	170					
6	AAA-3-D	2.44	3.32	3.7			2.30								0.00	1460					
7	AAA-3-D	3.32	3.66	3.9			6.40								0.00	6480					
8	AAA-3-D	3.66	4.57	4.1			3.07								0.00	2190					
9	AA-2-AH	0.00	0.46	4.0	31.0		2.93	32.10					0	9.1	2.10	1930	3320	2820	2990	410	9.4
10	AA-2-AH	0.46	0.91	4.0	31.0		2.05	27.10					0	8.5	3.00	1270	2700	1730	3080	370	10.5
11	AA-2-AH	0.91	1.22	4.0	19.0		1.19	8.49	10	59	31	SiCL	0	5.3	1.10	570	1940	3710	4060	340	7.0
12	AA-2-D	1.22	1.46		16.0			16.40							0.00	0	2740	2120	3620	300	1.7
13	AA-2-D	1.46	2.01	3.2	14.0		5.21	5.87	10	63	27	SiCL	0	3.0	27.00	4570	1880	2200	4430	300	1.3
14	AA-2-D	2.01	2.44	3.7	6.8		5.72	2.42	41	49	10	L	0	6.0	26.00	5370	1860	2540	7890	350	2.3
15	AA-2-D	2.44	?	2.1	<1.5		30.5	14.50	59	32	9	SL	0		< 0.1	0	770	2330	4760	300	<1.0
16	AA-2-D	3.41	4.27	3.9	3.8		5.90	2.08	77	20	3	LS	0	10.0	< 0.1	5250	650	2710	5980	330	28.6
17	AA-3-CORE	0.43	1.13	4.2			0.84									280					
18	A-3-AH	0.00	0.15													0					
19	A-3-AH	0.15	0.46	4.8			3.25									2520					
20	A-3-AH	0.46	0.76	3.9			1.93									1030					
21	A-3-AH	0.76	0.85	4.2			1.58									880					
22	A-3-D	0.88	1.04	3.7			2.01									1210					
23	A-3-D	1.04	1.95	4.2			0.57									190					
24	A-3-D	1.95	2.44	3.7			3.95									3230					
25	B-2-AH	0.00	0.21	3.5			1.64									719					
26	B-2-AH	0.21	0.61	2.0			20.10									36500					
27	B-2-AH	0.61	1.22	3.2			5.20									4600					
28	B-2-D	1.46	1.77													0					
29	B-2-D	1.77	1.95	3.5			5.25									5720					
30	B-2-D	1.95	2.07	4.2			1.59									710					
31	B-2-D	2.07	2.44	3.6			9.52									14100					
32	B-2-D	2.44	3.66	3.4			7.98									11100					
33	B-3-CORE	0.00	0.52	4.6			0.00									0					
34	B-3-CORE	0.52	1.13	3.6			0.47									100					
35	B-3-CORE	1.13	2.13	4.4			1.15									550					
36	B-4-CORE	0.00	0.30	6.6			0.00									0					

Appendix C-1. cont.

NO	SITE	DEPTH HIGH m	DEPTH LOW m	pH	CEC meq/ 100g	AEC meq/ 100g	EC mmhos /cm	ORGAN S CARB %	Si C % %	TEXT URE	HCO3 Cl mg/l	F ppm	SO4 mg/l	Ca ppm	K ppm	Mg ppm	Na ppm	P ppm
37	B-4-CORE	0.61	0.98	5.0			3.09						2140					
38	B-4-CORE	0.98	1.65	4.3			2.15						1440					
39	B-4-CORE	1.65	1.80	5.2			1.42						630					
40	BC-1-AH	0.00	0.46	4.1			0.94						450					
41	BC-1-AH	0.85	1.22	3.7			3.51						2810					
42	BC-1-2-AH	0.00	0.30	3.2			1.70						880					
43	BC-1-2-AH	0.30	1.04	4.0			1.59						830					
44	BC-1-2-AH	1.04	1.22	4.3			0.00						0					
45	C-1-AH	0.00	0.51	2.6	4.9	90	3.22	12.40	37	50	13	5.4	2030	370	630	370	270	<1.0
46	C-1-AH	0.91	1.22	4.1	14.0	28	2.42	10.20	29	52	19	6.7	1500	970	1020	800	270	<1.0
47	C-1-D	2.38	3.08	3.7	28.0	18	3.61	14.80	19	48	33	4.7	2850	3450	1910	3570	360	4.2
48	C-1-D	3.08	3.23		9.2		0.00	5.81	18	71	11	5.8	0	1710	1900	3220	580	8.6
49	C-1-D	3.23	3.44		29.0		0.00	28.10					0	9100	700	890	350	1.6
50	C-1-D	3.44	3.69	2.8	13.0		10.50	18.30	37	57	6	0.18	20500	5150	2020	3240	390	<1.0
51	C-1-D	5.21	5.24	4.7	4.4		3.61	1.33	75	18	7	2.40	0	1830	2400	8790	340	6.3
52	C-1.5-AH	0.00	0.30	3.7			3.08						1790					
53	C-1.5-AH	0.30	0.61	4.5			3.71						2750					
54	C-1.5-AH	0.61	1.22	3.5			0.36						30					
55	C-2-AH	0.00	0.15				0.00						0					
56	C-2-AH	0.15	0.61	3.5			5.23						4750					
57	C-2-AH	0.61	1.04	4.0			1.42						780					
58	C-2-AH	1.04	1.22	3.9			0.34						48					
59	C-2-D	2.13	2.80	2.9			8.31						11200					
60	C-2-D	2.80	3.44	2.5			14.50						28800					
61	C-2-D	3.44	3.75	4.9			1.43						590					
62	C-3-AH	0.00	0.30	4.1			4.11						3900					
63	C-3-AH	0.30	1.13	4.2			0.33						100					
64	C-3-D	1.22	1.52	4.0			0.40						80					
65	C-3-D	1.52	?	4.9			1.12						470					
66	C-4-AH	0.00	0.46	6.2	45.0		1.54	21.60	22	59	19	7.3	730	2020	5600	9330	360	2.9
67	C-4-AH	0.46	0.91	4.5	12.0		1.58	6.10	15	62	23	0.33	820	7200	5140	7780	300	7.4
68	C-4-AH	0.91	1.22	4.1	9.8		0.51	3.46	25	56	19	0.17	225	8700	4420	6440	370	15.9
69	C-4-D	0.46	1.86	3.8	12.0		3.08	5.37	31	50	19	4.3	2420	1610	5100	7470	360	9.5
70	C-4-D	1.86	3.11	5.1	3.2		1.99	1.15	67	26	7	1.9	1250	1030	3090	8540	310	7.9
71	C-5-AH	0.00	0.21	5.3			3.63						2940					
72	C-5-AH	0.21	0.43	3.9			2.91						2160					

Appendix C-1. cont.

NO	SITE	DEPTH HIGH m	DEPTH LOW m	PH	CEC meq/ 100g	AEC meq/ 100g	EC mmhos /cm	ORGAN S CARB %	Si C %	TEXT URE	HCO3 Cl mg/l	F mg/l	SO4 mg/l	Ca ppm	K ppm	Mg ppm	Na ppm	P ppm
73	C-5-D	0.43	?	4.7			0.00						0					
74	C-5-D	1.22	1.98	4.9			0.76						320					
75	C-5-D	1.98	?	5.6			0.52						150					
76	C-6-CORE	0.00	0.21				0.00						0					
77	C-6-CORE	0.21	0.61	5.3			0.38						100					
78	C-6-CORE	0.91	1.31	5.9			0.45						0					
79	CD-1-AH	0.00	0.40	3.3			2.52						1610					
80	CD-1-AH	0.40	0.61	4.3			2.29						1500					
81	CD-1-AH	0.61	1.22	4.5			0.41						120					
82	CD-1-2-AH	0.00	0.40	3.2			1.69						970					
83	CD-1-2-AH	0.40	0.61	4.2			2.28						1530					
84	CD-1-2-AH	0.61	1.22	3.9			3.77						3280					
85	D-1-AH	0.00	0.30	4.4			2.21						1340					
86	D-1-AH	0.30	0.55	4.5			2.39						1660					
87	D-1-AH	0.55	1.22	4.4			0.33						74					
88	D-1	0.98	?	4.7			1.15						570					
89	D-1	3.11	4.27	4.0			5.79						5900					
90	D-2-AH	0.00	0.30	5.2			3.22						2380					
91	D-2-AH	0.30	0.91	4.5			1.18						600					
92	D-2-AH	0.91	1.22	4.5			0.22						34					
93	D-2-D	2.32	2.44	5.3			0.00						0					
94	D-2-D	2.50	2.99	3.5			3.43						2920					
95	D-3-AH	0.00	0.09				0.00						0					
96	D-3-AH	0.09	0.30	4.1			3.99						3680					
97	D-3-AH	0.30	0.46	4.3			0.64						190					
98	D-3-AH	0.46	0.61	4.1			1.45						750					
99	D-3-AH	0.61	1.22	4.1			0.35						97					
100	D-3-CORE	2.01	2.44	5.3			1.41						670					
101	E-1-AH	0.00	0.12	4.3			0.00						0					
102	E-1-AH	0.12	0.61	4.2			3.85						3230					
103	E-1-AH	0.61	1.22	4.4			1.58						1820					
104	E-1-CORE	2.59	2.93	4.2			0.00						0					
105	E-1-CORE	2.83	2.93	4.1			3.52						2580					
106	E-1-CORE	2.93	3.02	2.9			0.00						0					
107	E-1-CORE	2.41	3.29	2.6			0.00						0					
108	E-1-CORE	3.29	3.66	4.8			4.19						3510					

Appendix C-1. cont.

NO	SITE	DEPTH HIGH LOW m	DEPTH LOW m	PH	CEC meq/ 100g	AEC meq/ 100g	EC mmhos /cm	ORGAN CARB %	S %	Si %	C %	TEXT	HCO3 mg/l	Cl mg/l	F ppm	SO4 mg/l	Ca ppm	K ppm	Mg ppm	Na ppm	P ppm	
109	E-2-AH	0.00	0.15	4.5			3.52															
110	E-2-AH	0.15	0.61	4.3			1.21															
111	E-2-AH	0.61	0.91	4.3			0.43															
112	E-2-AH	0.91	1.22	4.3			0.24															
113	E-2-CORE	0.82	1.65	4.3			0.21															
114	E-2-CORE	1.65	2.01	4.6			1.01															
115	E-2-CORE	2.01	2.68	4.2			1.17															
116	E-2-CORE	2.68	3.35	4.0			1.52															
117	E-2-CORE	3.35	3.66	4.8			3.19															
118	E-3-CORE	0.00	0.24	5.5			3.11															
119	E-3-AH	0.24	0.49	4.3			3.61															
121	F-1-AH	0.00	0.09		38.0		1.77	33.00	10	57	33	SiCL	0	11.0	0.3		1720	2290	1030	350	12.0	
122	F-1-AH	0.09	0.61	4.6			0.24	16.40	10	53	37	SiCL	0	3.9	<0.1		870	2860	1980	3280	300	1.6
123	F-1-AH	0.61	1.22	4.4			0.56	6.28	10	53	41	SiCL	0	5.4	<0.1		26	2100	2850	4670	320	2.6
124	F-1	1.22	1.77	4.3			0.00	5.58	4	55	41	SiCL	0				180	2750	3200	5170	330	2.2
125	F-1	1.77	1.89	4.5			0.00	4.66	12	49	39	SiCL	0				0	1900	2080	5220	290	3.6
126	F-1	1.89	2.26	3.6			5.42	6.04	24	55	21	SiL	0	2.3	0.19		5010	2620	2420	6170	280	3.7
127	F-1	2.26	2.41	4.0			2.11	4.54	23	49	28	CL	0	4.2	0.29		1220	2520	3160	6020	310	2.8
128	F-1	2.41	2.80	3.9			2.39	4.44	21	57	22	SiL	0	6.1	0.57		1470	2180	2820	5820	310	3.8
129	F-1	2.80	3.11	5.0			4.82	1.73	71	25	4	SL	0	13.0	<0.1		4040	3970	3410	9800	320	2.6
130	F-2-AH	0.00	0.15	5.3			3.61										2760					
131	F-2-AH	0.15	0.76	5.0			1.38										640					
132	F-3-AH	0.00	0.27	6.1			2.79										1770					
133	F-3-AH	0.27	0.52	6.2			0.71										290					
134	F-3-D	0.00	?	6.7			0.69										210					
135	F-3-D	1.22	1.83	6.7			0.62										225					
136	SEEP	0.00	0.27	3.1	6.7		2.67	4.34	67	24	9	SL	0	8.0	3.5		2010	2790	4090	4620	580	6.5
137	SEEP	0.27	1.22	3.9	11.0		1.78	3.72	50	33	17	L	0	5.7	1.1		1040	3370	4360	5230	430	7.7
138	SEEP	1.22	2.10	3.8	8.6		2.17	3.38	56	33	11	SL	0		1.3		1430	4720	4820	5640	470	6.0
139	SEEP	2.10	2.44	5.1	10.0		1.58	2.12	70	21	9	SL	22.5	6.9	<0.1		870	6200	3750	5250	400	6.7
140	GRAY TAILS	0.00	0.61	2.9	<1.5		17.00	6.10	16	53	31	SiCL	0		0.35		24300	6200	2990	1680	240	<1.0
D6	AAA-3-D	2.44	3.25	3.7			2.30										1410					
D15	AA-2-D	2.44	?	2.1	<1.5		32.00	13.70	59	31	10	SL	0		<0.1		0	760	2350	4740	290	<1.0
D20	A-3-AH	0.46	0.76	3.9			2.20										1230					
D32	B-2-D	2.44	3.66	3.5			7.87										11300					
D40	BC-1-AH	0.00	0.46	4.1			1.10										450					

Appendix C-1. cont.

NO	SITE	DEPTH HIGH	DEPTH LOW	DEPTH m	PH	CEC meq/ 100g	AEC meq/ 100g	EC mmhos /cm	ORGAN CARB %	S	Si %	C %	TEXT URE	HCO3 mg/l	Cl mg/l	F ppm	SO4 mg/l	Ca ppm	K ppm	Mg ppm	Na ppm	P ppm
D54	C-1.5-AH	0.61	1.22	3.9				0.30														
D-60	C-2-D	2.80	3.44	2.5				14.70														
D63	C-3-AH	0.30	1.13	4.1				0.26														
D70	C-4-D	1.86	3.11	5.2		3.2		2.00	1.05	65	26	9	SiL	0		0.20	1210	990	3010	8850	270	7.2
D87	D-1-AH	0.55	1.22	4.4				0.35														
D89	D-3-AH	0.61	1.22	4.0				0.31														
D110	E-2-AH	0.15	0.61	4.3				1.20														
D123	F-1-AH	0.61	1.22	4.3		23.0		0.52	6.00	10	53	37	SiCL	0	4.1	<0.1	24	2060	2810	4590	310	
D140	GRAY TAILS	0.00	0.61	2.9		<1.5		17.00	5.98	14	52	34	SiCL	0		0.33	24000	6550	3020	1700	270	

Appendix C-2. Swamp Gulch wetland sediment metals data (ppm).

NO.	Site	Depth High	Depth Low	Ag	Al	As	Ba	Be	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Sb	Sn	Tl	Zn
1	AAA-3-AH	0.00	0.24		14600						210	51500		160		450		<10		1780
2	AAA-3-AH	0.24	0.61		25900						160	32700		130		135		<10		170
3	AAA-3-AH	0.19	1.22		28300						180	36900		140		155		<10		160
4	AAA-3-D	1.46	2.13		19500						150	24200		130		130		<10		200
5	AAA-3-D	2.13	2.44		23700						140	34400		140		155		<10		150
6	AAA-3-D	2.44	3.32		25600						220	24300		140		145		<10		380
7	AAA-3-D	3.32	3.66		20200						3270	21300		170		165		<10		1470
8	AAA-3-D	3.66	4.57		19400						1250	24700		110		90		<10		220
9	AA-2-AH	0.00	0.46	0.1	10900	18.5	170	17.0	2.0	<5	44	36400	<1	140	9.0	79	<1	<10	<1	150
10	AA-2-AH	0.46	0.91	1.5	15600	35.0	680	12.0	1.7	<5	140	25900	<1	130	7.8	100	<1	<10	<1	53
11	AA-2-AH	0.91	1.22	3.0	29200	35.0	1230	10.0	1.5	8.5	130	20900	<1	110	6.5	130	<1	<10	<1	73
12	AA-2-D	1.22	1.46	1.7	18900	33.0	990	12.0	1.8	<5	150	26200	<1	150	9.6	115	<1	<10	<1	91
13	AA-2-D	1.46	2.01	2.3	23900	26.0	650	10.0	1.9	<5	150	21500	<1	130	11	135	<1	<10	<1	110
14	AA-2-D	2.01	2.44	0.9	21900	20.0	590	9.9	1.8	<5	89	20300	<1	170	22	100	<1	<10	<1	140
15	AA-2-D	2.44	?	1.1	30500	32.0	500	21.0	8.5	<5	370	43700	<1	64	130	75	<1	<10	<1	2300
16	AA-2-D	3.41	4.27	<0.5	21200	17.0	460	9.0	7.2	<5	460	18500	<1	110	47	60	<1	<10	<1	1170
17	AA-3-CORE	0.43	1.13		11200						8900	99700		100		70		<10		140
18	A-3-AH	0.00	0.15		6310						52	124000		500		245		<10		4080
19	A-3-AH	0.15	0.46		11600						41	59500		120		95		<10		1130
20	A-3-AH	0.46	0.76		20600						57	54000		120		115		<10		400
21	A-3-AH	0.76	0.85		24300						32	75700		84		115		<10		200
22	A-3-D	0.88	1.04		22900						76	55500		120		120		<10		250
23	A-3-D	1.04	1.95		19900						59	36200		120		75		<10		150
24	A-3-D	1.95	2.44		21200						250	14600		86		95		<10		1150
25	B-2-AH	0.00	0.21		3320						55	264000		<1		25		<10		61
26	B-2-AH	0.21	0.61		8370						8	100300		49		26		<10		260
27	B-2-AH	0.61	1.22		24500						110	108000		100		66		<10		140
28	B-2-D	1.46	1.77		22000						47	118000		60		32		<10		450
29	B-2-D	1.77	1.95		46300						200	54000		60		56		<10		940
30	B-2-D	1.95	2.07		58000						360	217000		<1		17		<10		2600
31	B-2-D	2.07	2.44		21900						1380	47800		29		87		<10		4900
32	B-2-D	2.44	3.66		16000						550	46300		89		42		<10		720
33	B-3-CORE	0.00	0.52		12600						40	213500		121		50		<10		210
34	B-3-CORE	0.52	1.13		26400						76	62500		140		92		<10		190
35	B-3-CORE	1.13	2.13		14100						100	28700		110		76		<10		180
36	B-4-CORE	0.00	0.30		19500						1000	40000		5250		3380		<10		3030

Appendix C-2. cont.

NO.	Site	Depth High	Depth Low	Ag	Al	As	Ba	Be	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Sb	Sn	Tl	Zn
		m	m																	
37	B-4-CORE	0.61	0.98		27000						660	35400		350		2500		<10		3150
38	B-4-CORE	0.98	1.65		28200						140	23300		150		200		<10		590
39	B-4-CORE	1.65	1.80		18000						45	26300		100		60		<10		310
40	BC-1-AH	0.00	0.46		34600						80	165700		53		68		<10		580
41	BC-1-AH	0.85	1.22		38200						420	22400		140		170		<10		170
42	BC-1-2-AH	0.00	0.30		9790						170	282000		290		130		<10		300
43	BC-1-2-AH	0.30	1.04		28400						140	284000		330		115		<10		190
44	BC-1-2-AH	1.04	1.22		61400						200	172400		250		165		<10		310
45	C-1-AH	0.00	0.91	5.0	2980	24.0	100	240.0	22.0	<5	930	415000	<1	<1	<1	530	<1	<10	<1	220
46	C-1-AH	0.91	1.22	13.0	4950	29.0	92	91.0	20.0	<5	770	157000	<1	150	25	1280	<1	<10	<1	1850
47	C-1-D	2.38	3.08		42000	35.0	810	35.0	14.0	<5	850	57000	<1	600	30	250	<1	<10	<1	770
48	C-1-D	3.08	3.23	1.1	10900	7.5	230	6.9	1.2	<5	75	11600	<1	99	5.0	24	<1	<10	<1	55
49	C-1-D	3.23	3.44	4.7	23000	22.5	210	45.0	21.0	<5	530	74900	<1	620	55	125	<1	<10	<1	1360
50	C-1-D	3.44	3.69	4.4	26800	23.5	530	47.0	10.0	<5	560	76000	<1	370	39	135	<1	<10	<1	700
51	C-1-D	5.21	5.24	0.9	16200	12.5	230	15.0	4.6	<5	580	25500	<1	130	23	280	<1	<10	<1	250
52	C-1.5-AH	0.00	0.30		16000						1940	286000		90		630		<10		3860
53	C-1.5-AH	0.30	0.61		24500						2700	135000		420		750		<10		11400
54	C-1.5-AH	0.61	1.22		39000						310	190000		480		140		<10		490
55	C-2-AH	0.00	0.15		26850						97	26500		96		185		<10		160
56	C-2-AH	0.15	0.61		33000						130	125000		190		115		<10		200
57	C-2-AH	0.61	1.04		6050						240	298000		110		420		<10		160
58	C-2-AH	1.04	1.22		19600						37	31600		100		115		<10		100
59	C-2-D	2.13	2.80		14300						1100	120000		280		180		<10		2030
60	C-2-D	2.80	3.44		25700						870	57000		210		290		<10		910
61	C-2-D	3.44	3.75		26500						650	157000		100		100		<10		1200
62	C-3-AH	0.00	0.30		25400						140	280000		82		80		<10		450
63	C-3-AH	0.30	1.13		31900						140	32700		120		75		<10		180
64	C-3-D	1.22	1.52		13200						760	45400		200		105		<10		250
65	C-3-D	1.52	?		25200						180	22000		920		125		<10		330
66	C-4-AH	0.00	0.46	1.9	27200	29.5	496	15.0	3.1	7.2	160	22000	<1	350	18	180	<1	<10	<1	310
67	C-4-AH	0.46	0.91	11.2	23100	86.0	910	29.0	20.0	<5	790	45200	<1	4740	21	3500	<1	<10	<1	2240
68	C-4-AH	0.91	1.22	9.6	25600	52.0	1140	47.0	18.0	<5	340	71000	<1	1630	65	2130	<1	<10	<1	3310
69	C-4-D	0.46	1.86	1.8	27600	22.0	580	13.0	2.0	7	66	21000	<1	120	14	900	<1	<10	<1	170
70	C-4-D	1.86	3.11	0.5	14000	11.0	370	5.7	<0.5	15	80	20600	<1	130	17	90	<1	<10	<1	170
71	C-5-AH	0.00	0.21		28800						740	51500		190		170		<10		750
72	C-5-AH	0.21	0.43		33600						130	32100		110		160		<10		170

Appendix C-2. cont.

NO.	Site	Depth High Low m	Ag	Al	As	Ba	Be	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Sb	Sn	Tl	Zn
73	C-5-D	0.43	?	31100						130	32000		150	150	195		<10		380
74	C-5-D	1.22	1.98	19500						250	20000		780	780	175		<10		270
75	C-5-D	0.00	?	14600						160	20800		310	310	90		<10		150
76	C-6-CORE	0.21	0.21	24000						200	22400		1640	1640	795		<10		620
77	C-6-CORE	0.21	0.61	48000						140	25600		420	420	250		<10		360
78	C-6-CORE	0.91	1.31	12700						21	12500		130	130	75		<10		110
79	CD-1-AH	0.00	0.40	9400						730	304000		42	42	510		<10		370
80	CD-1-AH	0.40	0.61	36500						2750	123500		630	630	750		<10		4250
81	CD-1-AH	0.61	1.22	57600						480	69000		960	960	510		<10		700
82	CD-1-2-AH	0.00	0.40	10900						1130	359000		80	80	660		<10		470
83	CD-1-2-AH	0.40	0.61	25200						3840	154000		260	260	2250		60		4490
84	CD-1-2-AH	0.61	1.22	57600						1520	37900		530	530	900		<10		2830
85	D-1-AH	0.00	0.30	18500						1030	252000		1180	1180	700		<10		2690
86	D-1-AH	0.30	0.55	46600						1360	62500		1550	1550	1500		80		2040
87	D-1-AH	0.55	1.22	48400						440	34100		460	460	360		<10		470
88	D-1	0.98	?	54300						570	34400		420	420	315		<10		630
89	D-1	3.11	4.27	19500						590	30000		170	170	130		<10		1180
90	D-2-AH	0.00	0.30	19800						2970	152000		420	420	1600		<10		6430
91	D-2-AH	0.30	0.91	45000						320	138000		790	790	60		<10		740
92	D-2-AH	0.91	1.22	34800						350	146000		570	570	150		<10		470
93	D-2-D	2.32	2.44	30500						260	22400		110	110	60		<10		250
94	D-2-D	2.50	2.99	41900						390	49100		240	240	170		<10		560
95	D-3-AH	0.00	0.09	8360						350	164000		900	900	1300		<10		2940
96	D-3-AH	0.09	0.30	21800						1000	78200		320	320	2050		<10		4400
97	D-3-AH	0.30	0.46	35700						240	49100		170	170	315		<10		330
98	D-3-AH	0.46	0.61	32800						230	98700		150	150	195		<10		210
99	D-3-AH	0.61	1.22	29500						140	46000		130	130	180		<10		170
100	D-3-CORE	2.01	2.44	12500						90	23700		120	120	95		<10		150
101	E-1-AH	0.00	0.12	7380						440	302000		270	270	2125		<10		620
102	E-1-AH	0.12	0.61	37400						1800	77200		860	860	300		<10		4520
103	E-1-AH	0.61	1.22	48300						510	37200		380	380	200		<10		510
104	E-1-CORE	2.59	2.83	23000						320	24200		160	160	110		<10		1350
105	E-1-CORE	2.83	2.93	36900						540	36800		250	250	195		<10		1020
106	E-1-CORE	2.93	3.02	23400						130	20900		70	70	70		<10		300
107	E-1-CORE	2.41	3.29	33700						550	74300		200	200	110		<10		2600
108	E-1-CORE	3.29	3.66	18700						5080	57500		260	260	255		<10		1050

Appendix C-2, cont.

NO.	Site	Depth High Low m	Ag	Al	As	Ba	Be	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Sb	Sn	Tl	Zn
109	E-2-AH	0.00	0.15	22100						1350	50000		520		165		<10		2690
110	E-2-AH	0.15	0.61	41000						260	89900		440		165		<10		650
111	E-2-AH	0.61	0.91	49300						280	107000		520		130		<10		470
112	E-2-AH	0.91	1.22	33000						350	76600		520		160		<10		440
113	E-2-CORE	0.82	1.55	28300						480	111400		690		260		<10		630
114	E-2-CORE	1.65	2.01	39000						520	34200		270		180		<10		320
115	E-2-CORE	2.01	2.68	27000						510	41600		270		140		<10		600
116	E-2-CORE	2.68	3.35	21800						350	30400		190		160		<10		340
117	E-2-CORE	3.35	3.66	14400						230	20500		160		100		<10		390
118	E-3-CORE	0.00	0.24	21600						910	63900		4940		2400		<10		2470
119	E-3-AH	0.24	0.49	20500						200	22100		250		300		<10		320
121	F-1-AH	0.00	0.09	8200	28.5	550	47.0	45.0	<5	1880	196800	<1	300	32	475	<1	<10	<1	2900
122	F-1-AH	0.09	0.61	35300	35.0	1160	19.0	44.0	14	1590	73700	<1	560	44	400	<1	<10	<1	2680
123	F-1-AH	0.61	1.22	43200	46.0	1950	27.0	4.0	18	530	106400	<1	290	26	170	<1	<10	<1	620
124	F-1	1.22	1.77	32400	34.0	1420	14.0	3.1	21	500	53900	<1	530	24	200	<1	<10	<1	560
125	F-1	1.77	1.89	32400	31.0	2140	31.0	<0.5	14	850	120800	<1	280	23	155	<1	<10	<1	440
126	F-1	1.89	2.26	29900	31.0	950	16.0	3.1	19	1110	57900	<1	1250	38	150	<1	<10	<1	860
127	F-1	2.26	2.41	36600	34.0	1420	20.0	<0.5	<5	590	78000	<1	790	26	200	<1	<10	<1	600
128	F-1	2.41	2.80	33800	31.0	1390	19.0	<0.5	18	790	71900	<1	440	29	140	<1	<10	<1	610
129	F-1	2.80	3.11	17800	18.0	840	12.0	<0.5	15	520	43000	<1	940	40	100	<1	<10	<1	650
130	F-2-AH	0.00	0.15	17100						440	44900		3200		1800		<10		1920
131	F-2-AH	0.15	0.76	26000						220	18800		300		245		<10		450
132	F-3-AH	0.00	0.27	18400						440	32500		2160		1050		<10		2040
133	F-3-AH	0.27	0.52	28500						181	28800		400		175		<10		440
134	F-3-D	0.00	?	19200						100	26200		250		84		<10		210
135	F-3-D	1.22	1.83	19000						120	27400		370		64		<10		210
136	SEEP	0.00	0.27	14.0	38.5	150	26.0	5.6	7.4	1050	116300	<1	550	19	8750	<1	<10	<1	1030
137	SEEP	0.27	1.22	30800	29.0	270	17.0	15.0	13	640	65800	<1	560	51	4000	<1	<10	<1	576
138	SEEP	1.22	2.10	27400	37.0	190	22.0	7.1	12	630	87200	<1	610	21	2700	<1	<10	<1	670
139	SEEP	2.10	2.44	23000	30.0	190	17.0	1.9	11	470	67000	<1	680	18	780	<1	<10	<1	380
140	GRAY TAILS	0.00	0.61	13400	70.0	52	25.0	7.4	61	1770	101300	<1	5060	55	5250	<1	<10	<1	1280
D6	AA-3-D	2.44	3.26	29500						220	25000		140		180		<10		380
D15	AA-2-D	2.44	?	1.1	26900	26.0	480	23.0	7.9	<5	380	44400	<1	61	130	<1	<10	<1	2360
D20	A-3-AH	0.46	0.76	20200						58	56500		84		55		<10		390
D32	B-2-D	2.44	3.66	17500						590	47400		45		75		<10		780
D40	BC-1-AH	0.00	0.46	34200						77	166200						<10		590

Appendix C-2. cont.

NO.	Site	Depth		Ag	Al	As	Ba	Be	Cd	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Sb	Sn	Tl	Zn
		High	Low																	
		m	m																	
D54	C-1.5-AH	0.51	1.22		33750							307	194000	481		140		<10		481
D-60	C-2-D	2.80	3.44		26400						870	55700		210		155		<10		910
D63	C-3-AH	0.30	1.13		32100						140	32500		120		185		<10		180
D70	C-4-D	1.86	3.11	0.5	13600	11.0	380	5.7	<0.5	70	14	21400	<1	120	17	85	<1	<10	<1	170
D87	D-1-AH	0.55	1.22		53200						420	34700		440		330		<10		490
D89	D-3-AH	0.51	1.22		26100						140	44600		130		170		<10		170
D110	E-2-AH	0.15	0.61		41900						260	88600		430		135		<10		610
D123	F-1-AH	0.51	1.22	1.6	42200	44.0	1870	26.0	3.8	17	510	103600	<1	290	19	160	<1	<10	<1	580
D140	GRAY TAILS	0.00	0.61	23.0	17700	74.0	57	24.0	6.9	64	1800	99300	<1	4910	51	4875	<1	<10	<1	1240

APPENDIX D

Precision and Accuracy of Vegetation Data, and Plant Species Observed at the Swamp Gulch Wetland

Appendix D-1. Accuracy and precision (at 90% confidence level) of vegetation data.

Element	Accuracy ¹	Precision ²
Al	44.5± 6.3%	± 4.5%
As	91.4±36.2%	± 5.6%
Cd	NC	±13.5%
Ca	85.8± 4.2%	± 2.3%
Cu	80.8± 5.8%	± 3.5%
Fe	60.0± 2.9%	± 5.5%
Pb	90.2±12.6%	± 9.4%
Mn	75.4± 4.2%	± 1.4%
Ni	NC	±65.6%
Zn	64.2±12.1%	± 7.9%

1 Based on the analysis of NBS reference samples.

2 Based on field replicate samples.

NC = Not calculated because the NBS value and the reference samples concentration were below the instrument detection limit.

Appendix D-2. Vascular plants and bryophytes of the Swamp Gulch
Wetland study site, 1987.

VASCULAR PLANTS

ASTERACEAE

Achillea millefolium L.
Agoseris aurantiaca (Hook.) Greene
Anaphalis margaritacea (L.) B. & H.
Aster junciformis Rydb.
Cirsium arvense (L.) Scop.
Erigeron peregrinus (Pursh) Greene
Hieracium scouleri Hook.
Senecio serra Hook.
Tragopogon dubius Scop.

BERBERIDACEAE

Berberis repens (Lindl.) G. Don

BETULACEAE

Alnus incana (L.) Moench
Betula glandulosa Michx. var *Hallii* (Howell) Hitchc.

BRASSICACEAE

Vicia americana Muhl.
Thapsia arvense L.

CARYOPHYLLACEAE

Cerastium berringianum Cham. & Schlecht.

CUPRESSACEAE

Juniperus communis L.

CYPERACEAE

Carex aquatilis Wahl.
Carex illota Bailey
Carex nebraskensis Dewey
Carex rostrata Stokes
Scirpus acutus Muhl.

ERICACEAE

Arctostaphylos uva-ursi (L.) Spreng.
Chimaphila umbellata (L.) Bart.
Ledum glandulosum Nutt.

Appendix D-2. Continued.

Pyrola asarifolia Michx.
Vaccinium sp.

EQUITACEAE

Equisetum sp.
Melilotus officinalis (L.) Pallas

JUNCACEAE

Juncus ensifolius Wikst.

ONAGRACEAE

Epilobium glaberrimum Barbey

PINACEAE

Abies grandis (Dongl.) Forbes
Abies lasiocarpa (Hook.) Nutt.
Picea engelmannii Parry ex Engelm.
Pinus contorta Dongl. ex Lond.
Pinus ponderosa Dongl. ex Laws. & Laws.
Pseudotsuga menziesii (Mirbel) Franco

POACEAE

Agrostis sp.
Bromus carinatus H & A
Bromus ciliatus L.
Danthonia unispicata (Thumb.) Munro ex Macoun
Elymus glaucus Buckl.
Elymus trachycaulus (Link) Gould ex Skinners.
Festuca idahoensis Elmer
Muhlenbergia andina (Nutt.) Hitchc.
Phleum pratense L.
Scolochloa festuacea (Willd.) Link.

ROSACEAE

Fragaria virginiana Miller
Geum macrophyllum Willd.
Rosa woodsii Lindl.

SALICACEAE

Populus tremuloides Michx.
Salix boothii Dorn

SPARGANIACEAE

Sparganium minimum Fries

NON-VASCULAR PLANTS

BRYOPHYTES

Brachythecium sp.

Cratoneuron filicinum (Hedw.) Spruce

Isopterygium pulchellum

Kopterygium pulchellum (Hedw.) Jaeg.

Leptobeyum pyriforme (Hedw.) Wils.

Polytrichum juniperinum Hedw.

Sphagnum tenellum Ehrh. ex Hoffm.